Environment Canada Environnement Canada

PROCEEDINGS OF A WORKSHOP ON THE DEVELOPMENT AND USE OF WATER QUALITY OBJECTIVES

TASK FORCE FOR WATER QUALITY GUIDELINES
CANADIAN COUNCIL OF MINISTERS OF THE ENVIRONMENT

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FOREWORD

This workshop was hosted by the Water Quality Objectives Division, Water Quality Branch, Environment Canada for the Task Force on Water Quality Guidelines of the Canadian Council of Ministers of the Environment.

The workshop was organized by the Water Quality Objectives Division and Donald D. Macdonald, MacDonald Environmental Sciences Limited, Vancouver, British Columbia.

The papers contained in this document express the views of the participants.

The following are definitions used by the CCME Task Force on Water Quality Guidelines:

Water Quality Guideline: numerical concentration or narrative statement recommended to support and maintain a designated water use.

Water Quality Objective: numerical concentration or narrative statement which has been established to support and protect the designated uses of water at a specified site.

PURPOSE OF WORKSHOP

The first Canadian Water Quality Guidelines were published in 1987 by the Canadian Council of Ministers of the Environment (CCREM 1987). The guidelines compiled the available information on the effects of physical, chemical and biological variables on the water quality of freshwater ecosystems in Canada. Recommendations, in the form of numerical limits or narrative statements, were provided for assessing water quality and for promoting the wise use of Canadian water resources.

The guidelines should not be regarded as inflexible values for national water quality, as water bodies across Canada vary widely in hardness, pH, rate of flow and many other characteristics. Many of these variables can affect the fate and effects of contaminants. Thus, in most cases, the general water quality guidelines cannot be used without amendment to take the local conditions into consideration.

The purpose of the workshop was to introduce some of the factors which have to be considered when using the water quality guidelines for regional or site-specific purposes.

Representatives from across Canada met for three days to hear invited speakers; to discuss regional activities; and to participate in a workshop designed to simulate an actual water quality objectives development exercise.

All speakers at the workshop were invited to prepare manuscripts on their presentations; these manuscripts are contained in these "Proceedings".

CCREM. 1987. Canadian Water Quality Guidelines. Prepared by the Task Force on Water Quality Guidelines of the Canadian Council of Resource and Environment Ministers, March 1987.

SUMMARY

The Task Force on Water Quality Guidelines of the Canadian Council of Ministers of the Environment requested the Water Quality Branch of Environment Canada to organize a workshop on the development and use of water quality objectives. The workshop was held in Halifax, Nova Scotia from September 19th to 21st, 1989.

The purpose of the workshop was two-fold:

1. To present and review papers on the development and use of water quality objectives (day 1 and 3), and
2. To simulate the negotiation of water quality objectives for a river flowing across the Canada-US border (day 2).

1. Highlights of Papers.

Site-specific water quality objectives may be required in many situations to make informed decisions on the management of aquatic resources. The Canadian Water Quality Guidelines are usually modified for the development of water quality objectives for specific sites or water bodies using local information on existing water quality, site-specific biological conditions, local hydrological patterns, and social and economic conditions that prevail in the study area. Some of the biological data can be obtained with laboratory bioassays using site water and organisms local to the area or in-situ bioassays to assess toxicity, or field toxicological studies.

A critical aspect of water quality objective setting is the analysis of exposure since physical, chemical and biological factors influence the fate of water quality variables, including toxic chemicals. Careful consideration must be given to the choice of transport and fate models because all of them have limitations - they are simulations of the natural environment.

Much of aquatic toxicology is laboratory based and its results must be carefully applied to the natural environment. Aquatic toxicology is challenged with improving its ability to provide realistic objectives, especially through the use of acute sublethal microscale tests, and practical microcosm-mesocosm systems.

Water quality objectives are usually based on assessments of single species and single-chemical experiments. Results from mesocosm studies can be used to confirm benchtop bioassays. Mesocosm experiments show that the nature of the response of an ecosystem cannot be predicted from these benchtop bioassays. The studies also indicated that the lack of a response at the ecosystem level, after an objective had been exceeded for a short time, did not necessarily demonstrate that the objective was lower than that required to protect the ecosystem over the long term.

Mutagenicity must be considered when establishing water quality objectives, to protect subsequent generations and also to protect present

generations from somatic mutations which have shown a propensity to develop into cancers. Quantitative estimates of "safe" levels of mutagenic chemicals require data derived from aquatic organisms exposed via the appropriate media. Such data are generally lacking, but these data gaps must be addressed, or appropriate estimates developed.

Water quality objectives have a number of uses and their consideration should be a component of any strategy developed to protect the environment from the effects of industrial effluents. Ideally, water quality objectives to protect aquatic life from toxic chemicals, particularly those which are persistent, should be specified before selecting control or pollution prevention strategies.

New approaches to water use management which focus on economic incentives, disincentives, and other market interventions by governments, must be initiated to complement the traditional protection/enforcement approach. Water quality objectives (to protect aquatic ecosystem health) must be set on the basis of long-term ecosystem needs - there should not be any attempt to second-guess the ability of society to meet the objectives in this objective-setting process.

Practical examples of the development of water quality objectives e.g. the international portion of the Saint John River, New Brunswick, illustrate the amount of preparatory work that must be invested. However this investment is repaid in the effective management of the Saint John River.

The Prairie Provinces Water Board water quality objectives consider unique biological, hydrological, geochemical and demographic characteristics of the river basins within its jurisdiction. The objectives are compatible with provincial objectives and can be used to identify potential interprovincial water quality concerns. The use of a two-level water quality objective can identify immediate water quality concerns related to the protection of downstream uses as well as distinguish long term trends in water quality.

The Province of Ontario uses the database CESARS (Chemical Evaluation Search and Retrieval System) in its assessment of potentially toxic chemicals. CESARS contains information on physico- chemical properties, environmental fate and toxicity for chemical substances. The database is available on CCINFO through the Canadian Centre for Occupational Health and Safety.

Quebec uses water quality objectives to develop environmental release objectives for point sources of water pollution. Regulations and monitoring programs are proposed and activities prescribed if corrective actions have to be taken to improve effluent quality.

The Saskatchewan Surface Water Quality Objectives were revised in November 1988, based mainly on the CCME Water Quality Guidelines. The revised Objectives include new guidelines for effluent mixing zones. Basin-specific water quality objectives are also being developed, the South Saskatchewan River basin is scheduled for completion in 1990.

Manitoba has been active in the development and use of water quality objectives based on a watershed classification system. There are four different levels of protection for aquatic systems denoted for Manitoba.

The development of water quality objectives for the St. Lawrence River has been included in the multi-agency St. Lawrence Action Plan designed to restore the river's quality and its beneficial uses. A list of priority substances has been established and site-specific water quality objectives will be developed for the different priority zones of interest in the river, such as Lake St. Francois and Lake St. Pierre.

2. Negotiation of Water Quality Objectives at the Canada-US Boundary.

The information given during the first day of papers, was put to use on the second day as the participants took part in a role playing exercise to develop water quality objectives for an international river. The workshop participants divided into four groups and the members acted as representatives of Canadian and US agencies developing water quality objectives.

The four groups role-played as follows:
1) Canadian officials working in isolation;
2) USA officials working in isolation;
3) Canadian officials who could consult with the USA officials, and
4) USA officials who could consult with the Canadian officials.

The participants used actual data from the international portion of the Flathead River flowing from British Columbia into Montana, USA.

Groups 3 and 4 demonstrated the benefits of cooperation, being able to exchange data and to discuss the significance of the data to the uses being made of the river by both countries. The exchange of information and close co-operation enabled groups 3 and 4 to propose water quality objectives for the river. The groups working in isolation were very restricted by the absence of data from the other side of the international border and had difficulty proposing water quality objectives.

During the workshop, the participants were able to learn about and discuss the various aspects (scientific and economic) of water quality objectives development and use. The exercise provided a useful opportunity for the participants to experience the mechanisms, and difficulties, of negotiating water quality objectives for a transboundary river.

ACKNOWLEDGEMENTS

The organisers wish to thank all who made the workshop a success especially those who presented papers and participated in the workshops.

Some Approaches to the Development of Water Quality Guidelines and Objectives

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INTRODUCTION

The development of water quality guidelines and objectives requires intensive use of a variety of types of information. Variables of concern are identified and recommendations are made for each variable. The recommendations may take the form of general guidelines, site-adapted guidelines, or site-specific guidelines. Any of these forms of guidelines are sometimes taken and designated as water quality objectives. This paper identifies major information needs involved in development of guidelines and objectives.

IDENTIFICATION OF VARIABLES

Variables requiring guideline development can be identified by examining water uses, existing water quality, and pollution factors. Specific water uses are known to be impaired by certain variables (e.g. fecal coliform bacteria are indicators of pathogens that impair drinking and recreational uses of water, metals are highly toxic to freshwater aquatic life, dissolved salts are a particular problem for irrigation uses, etc.). Study of "background" water quality may reveal variables that are already out of normal ranges or indicate conditions that will create additional or interactive stresses with pollutants. Pollution factors themselves indicate the choice of variables for defining objectives in a water body.

Often, neither "background" water quality nor pollution factors are well characterized in a water body. Thus there is a requirement for special studies and monitoring programs to identify the variables of concern for development of guidelines.

GENERAL GUIDELINES

General water quality guidelines for the identified variables may be used as a starting point for water quality objectives. Commonly used existing guidelines are CCREM 1987, EPA 1986, and others. However, guideline documents are highly abstracted and sometimes out of date relative to the open scientific literature. Thus the original detailed toxicology literature is often consulted to supplement guideline documents. Toxicology bibliographic data bases, such as the EPA funded "Aquire," are useful in this regard.

Information contained in guideline documents and the toxicology literature may not be sufficient to specify guidelines or objectives, so that new toxicology information must be developed. Often, additional information is required on sublethal chronic effects, on how toxicity of a material varies over ranges of environmental variables, and on joint toxicity of pollutants. When there is no opportunity to obtain this information, guidelines are obtained by using available LC50 information and application and safety factors.

SITE-ADAPTED GUIDELINES

Because conditions in specific aquatic environments can differ substantially from those of laboratory bioassays, general guidelines are often modified for development of water quality objectives. Modifications are only possible for a few variables with known dependencies on other ambient conditions (e.g. ammonia with pH and temperature, metals with hardness). Little is known about how the toxicity of organic pollutants varies with other environmental variables. For cases where modifications of general guidelines can be made, it is important to have good information on existing water quality at the site, again from monitoring and special studies.

Site-adapted guidelines are also determined by what is known or can be learned about the different species present, most susceptible life stages present, and timing of presence. An example of a detailed analysis of this kind is MacDonald *et al.*'s (1987) site-specific water quality criteria for nitrate, nitrite, and ammonia for aquatic life in the Canadian Flathead River. Knowledge of species present and distribution and timing of life stages allows very detailed definitions of water quality requirements and monitoring strategies for these nitrogen compounds for specific sites and times of years.

SITE-SPECIFIC GUIDELINES

The distinction between site-adapted and site-specific guidelines is a difficult one to define, since the degree of site specificity incorporated in a water quality guideline increases gradually as more local considerations are taken into account. However, site-specific guidelines are usually understood to incorporate some *in situ* cause-effect work. New bioassays may be conducted to test toxicity under a site-specific set of conditions. Bioassays may be performed on the species of fish or other organisms found in the site. Site water may be used in bioassay work (see Willingham 1988).

It should be noted that guidelines may also be based on *in situ* bioassay responses (e.g. no Daphnid reproductive effects) rather than only on concentrations of chemicals in various media. In addition, *in situ* cause-effect experimental work may also be used to determine deleterious levels of exposure to suspended sediments (concentration/duration combinations) or nutrient regimes causing undesirable primary production conditions. Such studies may involve the use of mesocosm experimental facilities, such as those employed by H. Mundie at Carnation Creek (Mundie, pers. comm.).

MONITORING FOR COMPLIANCE WITH OBJECTIVES

Another area requiring further development is compliance monitoring. At present, it is common practice to use whatever monitoring data exist to compare with guideline or objective levels as a means of determining compliance. However, most or all monitoring programs were not designed for exceedance detection and are therefore not suited for this purpose.

Designing monitoring for compliance with water quality objectives is an integral part of the objective formulation process, since the monitoring must reflect the variables, distribution, and timing of the objectives themselves. New approaches are suggested (Whitfield 1988, Valiela and Whitfield 1989 and in press) but much work remains to be done in this field.

CONCLUSIONS

Development of general water quality guidelines, site-adapted guidelines, and site-specific guidelines requires intensive use of many types of information. General water quality guideline documents, bibliographic toxicology data bases and the original scientific literature all provide a good starting point for this work. Special studies of water quality and monitoring activity are usually required to understand existing water quality and the qualitative and quantitative nature of point and diffuse source pollution.

There is an urgent need to develop more toxicological information, especially on sublethal effects of environmental pollutants. Other important areas requiring development include dependencies of toxicity of pollutants on other environmental variables and joint toxicity of different pollutants. Information is also required on other aquatic ecosystem responses to environmental perturbations. For example, the relationship between nutrients and stream eutrophication is currently poorly understood. Also, the relevance of research in quantitative structure/activity relationships (QSAR) to water quality objectives development is currently unknown, but may prove useful for management purposes. This possibility ought to be explored.

Field toxicology is of great importance in the development of water quality objectives. A few research initiatives in this area should be supported, augmented and applied to water management. As well, new approaches are required for monitoring for compliance with objectives.

Thus it is apparent that development of water quality guidelines and objectives requires expanded efforts in research and applied science. In spite of the magnitude of these information requirements, progress can be made through cooperation and communication. Resources would be most efficiently and effectively utilized in joint and complementary work among federal, provincial, and municipal governments, the private sector, universities, and other non-governmental organizations.

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*These references have a series of volumes for different pollutants.

Hazard Assessment
The use of fate models to assess exposures of aquatic organisms to environmental contamination

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INTRODUCTION

For many, hazard assessment is synonymous with toxicity testing. This perception has persisted through the years, in part, because regulatory agencies and environmental quality managers have theoretically embraced a conservative approach to the protection of the quality of air, water, sediments and biota. This approach was perhaps best exemplified by the passage of the Delaney amendment to food and drug legislation in the United States in 1958, which specified that no chemical additive shown to cause cancer in laboratory animals could be added to food. Another example of a conservative approach is banning the production or use of a compound.

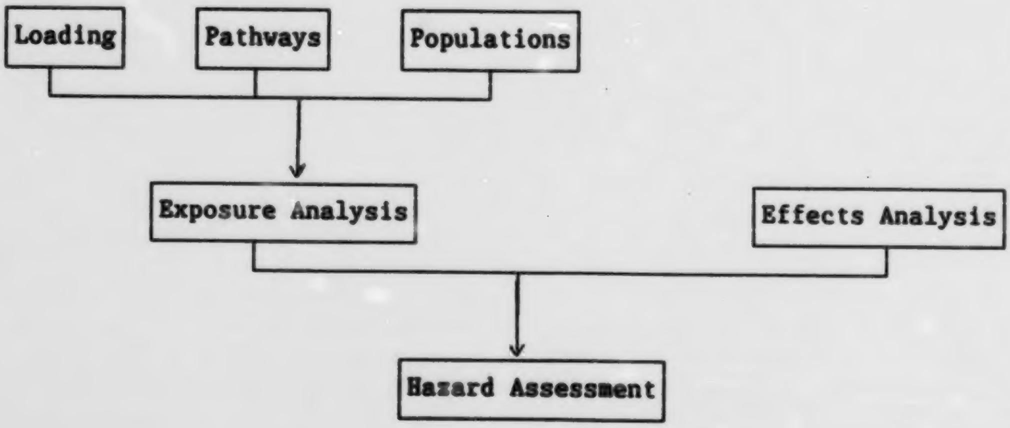
In both instances, there have been very few actions taken by regulatory agencies in the environmental protection, health protection, consumer protection or occupational health and safety fields. This has occurred even though there are approximately 100 000 commercial chemicals in use today in several hundred thousand formulations. Thus decisions will necessarily continue to be made in the face of uncertainty. It has been recognized that more rational and systematic schemes are needed for evaluating the hazard associated with chemicals, because we are dealing with degrees of risk.

LLoyd (1981) has stated that "water quality standards for aquatic life protection are required only for those chemicals which are likely to occur in harmful concentrations within the aquatic environment". That sounds like a truism to many of us but his statement emphasizes two critical points. The first is occurrence and the second is harmful concentrations. Lloyd has highlighted the two elements of hazard assessment: exposure and effects (Figure 1).

We know, as a result of experience that while a chemical may exhibit some toxic properties, exposure in the environment may not necessarily occur or that when exposure does occur, the levels are much lower than those causing lethal or sub lethal responses (if the latter are known). As stated by Howard et al. (1978) "It is quite, conceivable that a highly toxic, readily degradable substance will cause less environmental damage than a less toxic, persistent chemical. For example, bis-chloromethyl ether is a potent carcinogen but it hydrolyzes to innocuous products in a matter of seconds". Another example can be drawn from the Northeastern New Brunswick Mine Water Quality Program of the early 1970's. Copper, lead and zinc were mined in northeastern New Brunswick and therefore concentrations of these metals were found in the effluents and the receiving waters. These waters were the nursery areas for an economically valuable recreational and commercial salmon fishery. Laboratory studies conducted in St. Andrews, New Brunswick and in Halifax identified the lethal levels of the metals individually and in combination, for juvenile Atlantic Salmon. But young salmon were found in the wild in waters with concentrations of metals in excess of these levels. Research subsequently found that both water hardness (as $CaCO_3$) and humic acid, naturally present in waters in the Maritimes, provided a degree of protection, essentially a safety factor. In the first instance, ions responsible for the hardness were found to interfere with the binding sites for the toxic metals while in the second, the humic acids complexed the metals rendering them unavailable. The result of this work, which admittedly only concerned lethality of juvenile _Salmo salar_ rather than ecosystem impacts, was a simple model based on some straightforward equations which assisted managers in determining whether effluents were likely to interfere with the water use objective (Cook and Côté 1972). While the regulatory agencies had to recognize that other chemicals might be present in the effluents that could cause other effects, the operation of the mines could continue with a level of treatment below that indicated by laboratory tests. These findings have forced us to realize the importance of transport and transformation processes in setting limits on the concentrations of a chemical or mixture of chemicals which ought to be measured in water or air. This, then is the other element of hazard assessment.

The management of chemicals, as a public policy issue, often appears to be led by developments in analytical capability and the resulting growth of information in the distribution of chemicals. Environmental groups, politicians and senior civil servants often respond quickly to reports that concentrations, some as low as a part per quadrillion, have been found in water, sediment or biota at a location. In the majority of cases, toxicological studies have never been conducted to determine the degree of concern that is reasonable, nor do we know whether finding the chemical in that location bears any relationship to other releases into other environmental conditions.

Figure 1. The Components of Hazard Assessment



The conservative approach as exemplified by the Delaney amendment, can be interpreted to mean 1) all chemicals released will result in exposure and as a consequence, in detrimental effects and 2) all chemicals should be tested under all possible environmental conditions before they are produced, used and disposed of. Unfortunately, that is too much to expect for new chemicals let alone all the existing chemicals in use today though significant changes have been made in testing requirements. In the case of existing chemicals, governments still rely to a degree on trial and error i.e., when a problem is noted, action is taken. However, a number of commercial chemicals are being subject to increasing scrutiny using structure-activity relationships, benchmark profiles and exposure and effects models to determine their likely fate in the environment. At that point detailed toxicological testing may be required.

While they address the marine environment, the "Guidelines for the Protection of the Marine Environment from Land Based Sources of Pollutants" published by the United Nations Environment Programme are very useful in placing fate studies into context. The framework contained therein as well as the accompanying text stress that a combination of environmental protection strategies will be necessary to maintain the environmental capacity of ecosystems. It should also be noted that the UNEP Guidelines recommend that a process which integrates information on actual and desired uses of an aquatic environment be combined with information on the fate and effects of pollutants to set quality objectives and standards (see Figure 2). The purpose of introducing the framework here is that it links hazard assessment to other elements of an aquatic environmental management program. Other relevant frameworks have been described by Wells and Côté (1988).

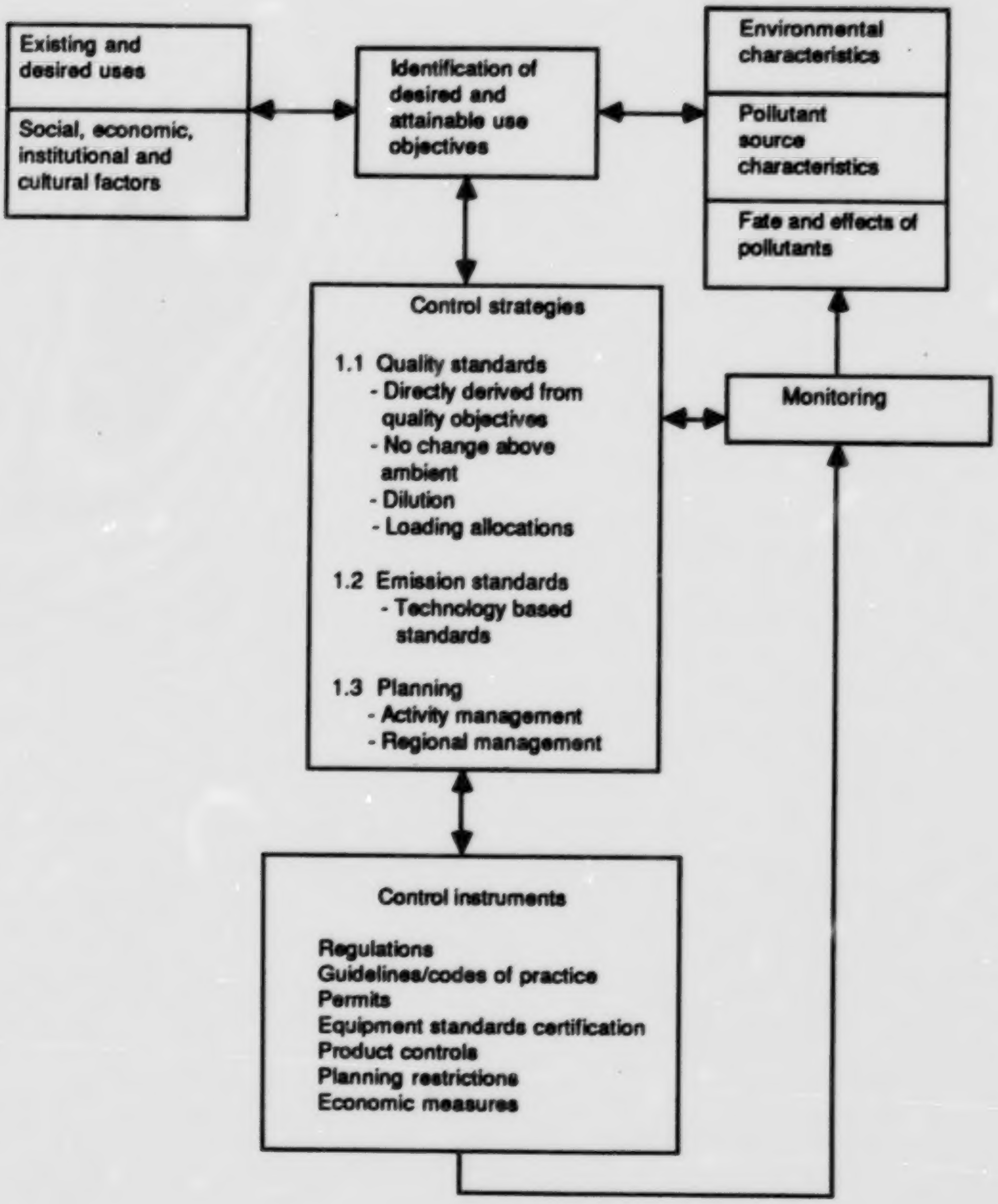
In summary, then, the reasons for emphasizing exposure in hazard assessment are 1) anthropogenic activities are causing increasing deterioration of surface and ground water 2) all uses of water may not be possible in perpetuity and objectives and criteria must be established which in turn determine the standards that will be applied and 3) the derivation of those objectives must recognize the physical and chemical properties of the chemicals and the environments into which they are released as these will influence whether a toxic effect will occur.

EXPOSURE ANALYSIS

The major exposure-related factors which influence toxicity of a chemical are route, duration and frequency. Understanding these three factors requires information on the identity of the compounds in question, their sources and emission rates, in addition to their concentrations in possible exposure media (air, surface, fresh water or sea water, groundwater, soil or sediment and biota).

Chemicals can be released in continuous discharges in industrial sewers or into the atmosphere and as spills. They can also be deposited into controlled or uncontrolled landfills. The majority of the studies reported in the literature on exposure and effects have investigated individual chemicals from a single source or a mixture of chemicals from a single source, eg. pulp and paper mill effluents. The most widely applied regulatory system in Canada is based on this approach. This system is

Figure 2. Marine Environmental Protection Framework as presented in UNEP Montreal Guidelines


Existing and desired uses
Social, economic, institutional and cultural factors
Identification of desired and attainable use objectives
Environmental characteristics
Pollutant source characteristics
Fate and effects of pollutants
Control strategies
1.1 Quality standards
- Directly derived from quality objectives
- No change above ambient
- Dilution
- Loading allocations
1.2 Emission standards
- Technology based standards
1.3 Planning
- Activity management
- Regional management
Monitoring
Control instruments
Regulations
Guidelines/codes of practice
Permits
Equipment standards certification
Product controls
Planning restrictions
Economic measures

technology-based at a best practicable technology (BPT) level. This is problematic because we are often dealing with multi-source, multi-exposure situations (see Figure 3). The water quality approach, which is the focus of this workshop, is a recognized alternative which has had limited application in this country except in the case of oxygen, nutrients and bacteria, but which has greater possibilities for addressing multiple source situations. But the task of developing objectives and criteria which are necessary to implement the water quality approach is an onerous one. In this regard, the most serious shortcoming is the limited research in water quality criteria and ecotoxicology.

After twenty-five years of environmental protection activity in this country, it should be clear that the nature of the pollutant and the environmental conditions into which those pollutants are discharged must play a large role in determining the level and type of treatment required. In situations where there is an individual effluent source on a reasonably large river, best practicable technology may be adequate to maintain most if not all uses. However, if multiple sources exist, the cumulative effects may exceed the capacity of the receiving water system. There will also be cases where chemicals are present which could bioconcentrate and accumulate in the ecosystem causing problems with particular biota; in such cases bans may have to be imposed on those chemicals or more substantive process changes may be required of the industries discharging into those waters. A case in point is the current situation with pulp and paper mills using the chlorine bleaching system.

We often forget that exposure can result at any and all points in the life cycle of a chemical (product R & D, production, transportation, storage, use and disposal) and the many products or uses of a chemical. Figure 3 from the report of the Commission on Lead in the Environment of the Royal Society highlights the many sources of that metal and potential exposure routes. Attention to polychlorinated biphenyls has focused on transformers and dielectric fluids but these compounds have been used in paints, varnishes, resins, pigments, adhesives, sealants, lubricants, plasticizers, pressure sensitive papers, fire retardents as well as heat transfer fluids since they were first manufactured in the early 1930's. Some percentage of the total PCBs used in Canada can be found in landfills, dumps and in sediments at the bottom of rivers and lakes. While effects or toxicity analyses are conducted primarily in the laboratory and can involve a wide variety of end-points (see Table 1), the exposure analysis is a study of the influence of the environment on the chemical, the pattern of chemical distribution and the organisms which could become exposed. The questions which should be answered in an exposure analysis are:

1. What chemicals or pollutants are present or likely to be present?

2. What are the sources of the chemicals?

3. Are these sources continuous, intermittent or accidental?

4. How does the chemical behave in media?

5. Does it bioaccumulate or biodegrade?

6. What are the mechanisms for change or removal in the media in question?

7. Does the chemical react with other compounds in the environment?

8. Is there transfer between media and what are the mechanisms and rates?

9. How long does the contaminant remain in the media?

10. What are the degradation products?

11. Is a steady state concentration achieved?

12. What is the resultant distribution or fate of the chemical?

13. Is this consistent between location, conditions, seasonally?

Where fate is now recognized as an important notion in the evaluation of hazard, "it has been equated to an observed spatial distribution, and not necessarily to concentrations nor to the factors responsible for distribution" (Lassiter *et al*. 1978). This has changed dramatically in the past ten years and is due in large measure to research on transport and transformation processes and improvements in modelling. This research has been prompted by regulatory initiatives in the United States (Table 2) and action of the Chemicals Group of the Organization for Economic Cooperation and Development.

Table 1. End points used in effects assessment

I. Individual species

- Short-term screening
  - Survival
  - Bioaccumulation
- Predictive-intermediate
  - Sublethal/integrative
  - Energetics/scope for growth
  - Genetics/pathology

  - Bioaccumulation
- Predictive long-term
  - Partial/complete life cycle
  - Survival, growth, reproduction
  - Bioaccumulation/biomagnification

II. Populations

- Population-demographic parameters
  - Intrinsic rate of growth
  - Reproduction value
  - Cohort analysis

III. Communities-multispecies

- Structural indices
  - Species dominance
  - Species diversity
  - Relative abundance
  - Species succession
- Functional indices
  - Biomass/productivity
  - Respiration

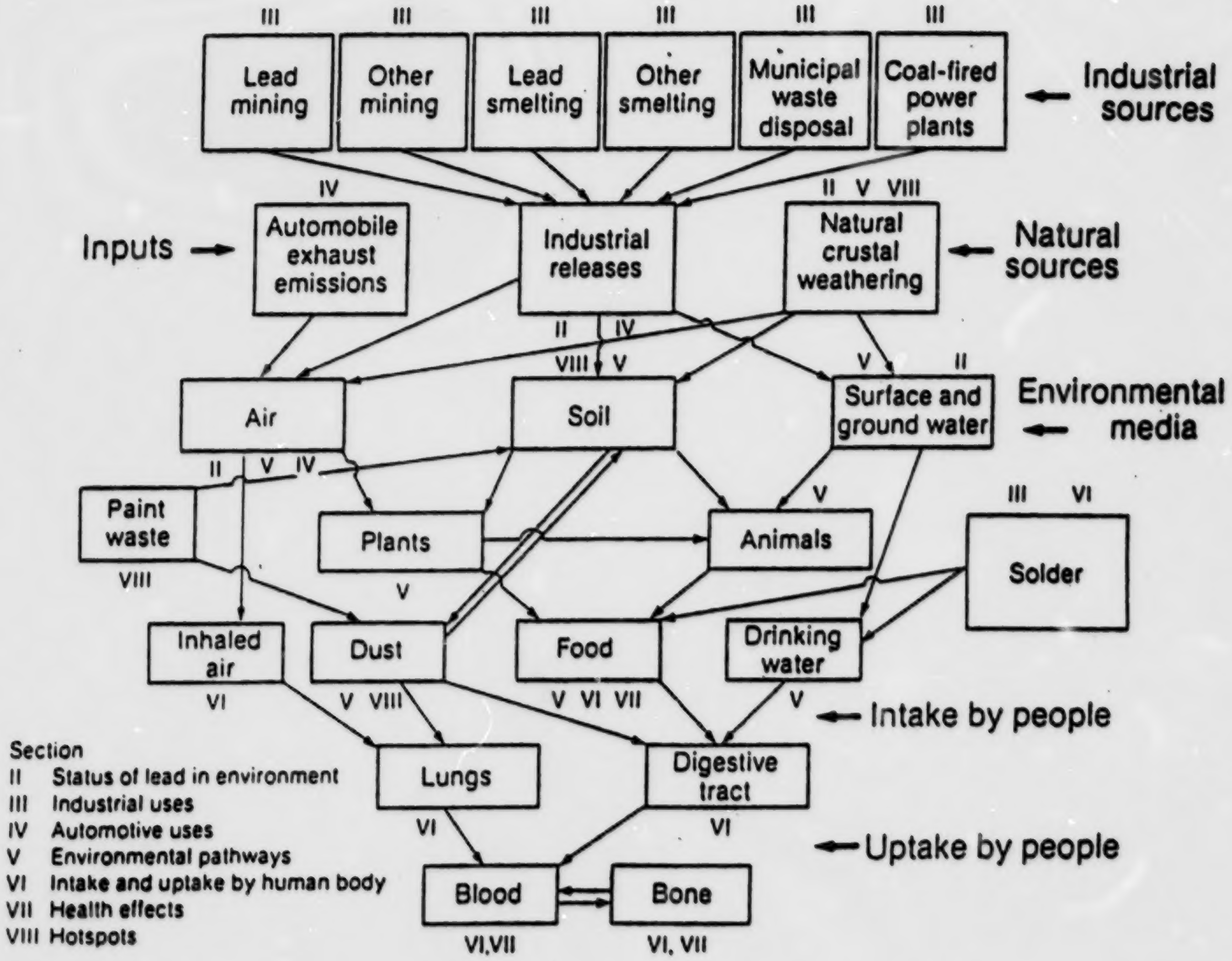


Figure 3. Sources and Pathways of Exposure to Lead in the Environment

From Commission on Lead in the Environment - Royal Society of Canada 1986

TRANSPORT AND TRANSFORMATION

The route, duration, magnitude and frequency of exposure are in turn influenced by transport and transformation process. Transport processes and/or measurements include:

Volatization
Solution
Adsorption
Desorption
Partitioning
Complexation
Acid leaching

The major transformation processes are:

Hydrolysis
Oxidation
Photolysis
Biodegradation

These factors and processes will determine whether exposure actually occurs (Howard et al. 1978). As indicated earlier some chemicals eg. bis-chloromethyl ether, will photolyse extremely rapidly and it is unlikely that environmental exposure would occur. This is not to say that occupational health concerns related to accidental releases of large quantities should be ignored. In another instance, a pollutant may have a high oxygen demand but the risk to aquatic species may only be significant during extreme low flow conditions in a river. In yet another situation, the chemical in question may adsorb strongly to sediment in a lake rendering it unavailable to fish until organic wastes build up on the bottom creating anoxic conditions resulting in methylation of the chemical, or until anoxic conditions are eliminated. (Office of Technology Assessment 1987.)

While we may be able to arrive at some conclusion regarding the distribution and fate of some pollutants conceptually, these have not proven to be very accurate in actual environmental situations. Numerical ranking systems and screening systems have been developed and used in many countries and industries. Unfortunately, there is always a risk that these systems will overestimate or underestimate the hazard associated with a chemical. If a regulatory agency or an industry underestimates or makes a decision by rejecting a prediction that will in fact occur (type 1 error) exposure may result and effects may be measured at some cost to all concerned. On the other hand, an overestimation or accepting a prediction that will not occur (type 2 error) will prevent exposure but may also deny an industry, and possibly society, of some benefit. In an attempt to reduce type 1 and type 2 errors models have become increasingly complex, often by combining relevant models and decision support expert systems.

Table 2. EPA Requirements for Exposure Analysis (US EPA 1984)

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1. General Information on the Chemical
   - molecular formula and structure
   - description of contaminants or additives
   - possible formulations
   - chemical and physical properties
     eg. boiling point, molecular weight, vapor pressure, partition coefficients, half-lives.

2. Sources
   - characterization of production and distribution
   - uses
   - disposal

3. Exposure pathways and environmental fate
   - transport
   - transformation
   - principal pathways of exposure
   - predicted environmental distribution

4. Estimated or monitored concentration levels
   - monitoring data
   - estimations of concentrations
   - comparisons of estimates and monitored concentrations

5. Exposed populations
   - human populations (size, location and habits)
   - other populations (size, location and habits)

6. Integrated exposure analysis
   - calculation of exposure
   - pathways of exposure
   - exposed scenarios
   - characterization of exposed population
   - evaluation of uncertainty

MODELS

Booty and Lam (1989) state that "the main thrust behind the development of water quality models has been the need to predict the results of man-made influences on our water resources, ranging from rainwater to seawater". From a very practical point of view Mackay and Paterson (1982) have expressed the view that "the environmental behaviour of chemicals is sufficiently complex that the human mind must be assisted by some form of systematic and physically reasonable mathematical model that described the partitioning, reaction and transport processes that interact to determine the exposure to which organisms are subjected."

Computer models, in particular, are viewed as "black boxes" by many professionals and managers and are in fact, limited representations of natural systems. As a result, decision-makers have had limited confidence in their predictions and output. The purpose for the model's use must be clearly understood and while it is not necessary for managers to understand all the mathematical calculations and the computer language involved, they should be made aware of the nature of the model, the assumptions upon which it has been constructed and its limitations. In this way, the unrealistic expectations of decision-makers might be reduced to a reasonable level. When output is generated, the uncertainties surrounding the data should also be clearly explained. Models do not replace managers; rather they are tools to aid managers in the decision-making process.

Models can be used in a number of ways. Some of these uses have a research orientation while others might be viewed as managerial.

Table 3. Uses of Aquatic Hazard Assessment Modelling

1. Testing hypotheses
2. Identifying knowledge gaps
3. Evaluating the significance of processes
4. Ranking chemicals for more detailed effects analysis
5. Testing water quality impact scenarios
6. Suggesting mitigation strategies
7. Teaching

There are various types of models and their categorization is problematic as a review of the literature has demonstrated. There are models based on experiments or monitoring programs conducted in actual field situations; single or multi-media models based on controlled laboratory studies; complex computer models constructed with information derived from a variety of sources in which the elements are entirely reduced to numbers, rate constants and equations and finally intelligent decision support systems which combine elements of the computer modelling approach with experts from different disciplines in workshop formats such as is utilized in the Adaptive Environment Assessment and Management approach. A further refinement is the linkage of biophysical models with an expert or knowledge based system which serves as an interface among the models. The expert system could aid in selecting inputs, explaining the model and interpreting outputs in terms of decisions (SRES 1989). Fate models range in complexity from simple estimations of distribution into various compartments of aquatic ecosystems which can be calculated by hand to sophisticated models requiring computers which estimate transport, transformation, rate of degradation and fate of the chemical under steady-state or dynamic conditions.

This paper emphasizes the application of computer models. These are normally categorized as conceptual relationships (laboratory tests), empirical stochastic (probabilities), or deterministic (steady-state or dynamic). Empirical models tend to be qualitative while stochastic and deterministic models are generally quantitative. As indicated by Stokoe *et al.* (1989) qualitative or semi-quantitative models "avoid the detail of values for variables and parameters and rely instead on information which is qualitative (is a species present or not?) or semi-quantitative (one process causes the concentration to increase while another process causes the opposite effect)".

Models are also described as:

(i) Equilibrium models: describe the idealized state of a system

(ii) Steady-state models: assume that the flux of a chemical or constituent into a compartment is equal to the flux out of that compartment or reservoir.

(iii) Time dependent models: are able to predict the consequences of a pollutant injection into the environment which changes over time.

(iv) Kinetic models: consider the distribution of a particular substance to be controlled by rate factors (can be either steady-state or time dependent).

Geographically, models are classed as:

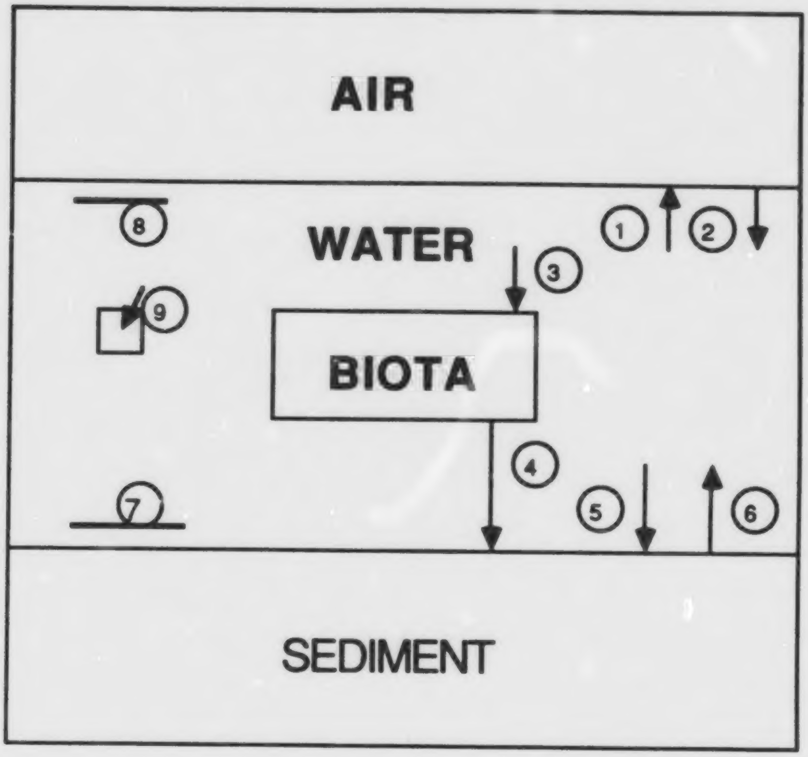
(i) Global scale
(ii) Mesoscale models (atmospheric, hydrologic)
(iii) Microscale

In a report for the Canadian Environmental Assessment Research Council, deBroissia (1986) categorized water quality models as:

(i) one dimensional - either vertical or horizontal - temperature or D.O.
(ii) two dimensional - along vertical or horizontal plane - salinity or contaminants
(iii) three dimensional - in all directions - temperature, D.O. contaminants
(iv) well-mixed - homogeneous - multicomponent
(v) hydrological - basin-wide - combination of models

Exposure analysis often involves compartment models of varying levels of complexity. A simplistic example is shown in Figure 4. The fugacity model (level I,II,III, and IV) and the Persistence Screening Model of the NRCC (Roberts *et al.* 1981) are compartment models.

Figure 4. A simple compartment model involving transport and transformation processes



① Volatization

② Fallout or deposition

③ Uptake (lipid solubility)

④ Excretion

⑤ Adsorption

⑥ Desorption

⑦ Biodegradation

⑧ Photolysis

⑨ Hydrolysis

A large number of models have been produced by university researchers, consultants, regulatory and water management agencies as attested by the incomplete list in Table 4. Many of the models were designed for very specific applications, focusing on particular groups of chemicals eg. pesticides, metals, polycyclic aromatic hydrocarbons, or specific environments eg. soil/groundwater linkages, groundwater, lakes, rivers,

Table 4. Some Models Available or in Current Use

ARM	Agriculture Runoff Model
NRCC	Persistence Screening Model (Roberts et al. 1981)
ALWAS	Air, Land, Water Analysis System (Cohen 1986)
EXAMS	Exposure Analysis Modelling System (Burns 1983)
Fugacity Model	Level I (Mackay and Paterson 1982; Cohen 1986)
	Level II
	Level III
	Level IV
MEXAMS	Metal Exposure Analysis Modelling System (Felmy et al. 1984)
MINTEQ	Program for Calculating Aqueous Geochemical Equilibria (Felmy et al. 1984)
PEST	Pesticides Transport in the Aquatic Environment (Burns 1983)
CLEAM	Comprehensive Lake Ecosystem Analysis Model (O'Neill 1982)
SRI	Standford Research Institute Watershed Model (Burns 1983)
SERATRA	Sediment-Radionuclide Transport Model (Burns 1983)
UTM-TOX	Unified Transport Model for Toxicants (Burns 1983; Cohen 1986)
SWRRB	Simulator for Water Resources on Rural Basins (Honeycutt and Ballantine 1983)
HSPF	Hydrologic Simulation Program - FORTRAN (Barnwell 1982)
IFEM	Integrated Fates and Effects Model (Bartell et al. 1988)
EECHEM	Exposure and Ecotoxicology Estimation for Environmental Chemicals (Brüggemann et al. 1987)
PRZM	Pesticide Root Zone Model
AT123D	Analytical Transient, One, Two, Three Dimensional Simulator of Water Transport in the Aquifer System (Donigian, Carsel 1987)
STREAM	Stream Transport and Agricultural Runoff of Pesticides for - Exposure Assessment (Donigian, Carsel 1987)
SLSA	Simplified Lake Systems Analysis (Games 1983)
SOPTRAN	Synthetic Oil Pollutant Transport Model (Herbes, Yeh 1985)
ADLITTLE	Multi Media Model (Eschenroeder 1981)
EXWAT	Chemical Transport Model for Surface Water Bodies (Brüggemann, et al. 1987)
CHEMEST	Chemical Property Estimation
QUALII	US. EPA Model for Rivers (deBroissia 1986)
LARM	Lakes and Reservoirs Model (thermal conditions) (deBroissia 1986)
ESTUAR	Modified estuaries model to predict salinity (deBroissia 1986)
RMA-3	Source-receptor relationship for sediment related contaminants (deBroissia 1986)
CEQUEAU	INRS - Eau model to calculate flow rates over time (deBroissia 1986)
WASP	Water Quality Analysis Simulation Program (DiToro et al. 1981)
SWACOM	Standard Water Column Analysis (O'Neill et al. 1982)

estuaries. However others are fairly generic in application. While it is not possible to provide an indication of the use of each of the models, several are known to have wide use and application. These include the fugacity model, EXAMS, SWACOM, the Hydrologic Simulation Program, the Integrated Fate and Effects Model and CEQUEAU.

The fugacity model developed by Mackay and Paterson (1982) available in four levels of complexity, consists of volumes of homogeneous air, water, soil, sediment, suspended matter and biota in which the distribution of the chemical is calculated from knowledge of physical and chemical properties. The key to this model is the concept of fugacity or the escaping tendency of a chemical from one phase into another until equilibrium is reached.

Level I - the partitioning of a fixed amount of a non-reacting compound is calculated using fugacity capacities based on physical and chemical data and partition coefficients.

Level II - the steady-state equilibrium concentrations of a chemical are calculated for fixed emissions that are balanced by reactions in each phase.

Level III - the chemical in question does not achieve equilibrium in all phases due to transfer resistances between phases.

Level IV - emissions or inflows into the system are not constant, changing with time. The output will be the time required for a chemical to accumulate to a given concentration in a compartment and the time for the system to recover when emissions are reduced.

EXAMS is an interactive program designed to assist in exposure evaluation of trace-level synthetic organic chemicals in the long term. It computes steady-state distribution of pollutant concentrations, the fate of pollutants in the system and the time required for effective purification of the system. A variant of this model is MEXAMS, a combination of features of EXAMS with MINTEQ which computes the aqueous speciation, adsorption and precipation/dissolution of metals (As, Cd, Cu, Pb, Zn, Ni, Ag) (Felmy et al. 1984).

SWACOM is a model which simulates the annual production cycle of the pelagic zone of a north temperate lake. It combines an environmental transport model with an effects matrix in order to predict the direct and indirect effects of a contaminant on predators, competitors and food organisms (Barnthouse et al. 1983).

The Hydrologic Simulation Program - FORTRAN (HSF) is a comprehensive simulation model using such information as rainfall, temperature, solar intensity, land use patterns, soil characteristics, agricultural practices and pesticide use to predict flow rate, sediment load and nutrient and pesticide concentrations at any point in the simulated watershed. This model goes further using a risk assessment methodology to evaluate lethal and sublethal effects (Barnwell 1982).

The Integrated Fate and Effects Model (IFEM) is a synthesis of two existing models FOAM and SWACOM, which predicts the distribution, fate and the effects of aromatic compounds on biological populations. It is designed such that there is feedback between the fates and effects of the toxicants (Bartell et al. 1988).

The CEQUEAU model developed by the Institut National pour la Recherche Scientifique has been applied to water quality studies in the Ste. Anne River and Yamaska River in Quebec. The HSP-F model has been used by several consulting firms in Canada to simulate water quality problems. In both cases, the models have been used to predict changes in BOD, temperature, phosphorous, nitrates, suspended solids (deBroissia 1986).

A useful review of fresh water quality models has recently been published by CCIW (Booty and Lam 1989). It summarizes characteristics for 38 nutrient and 35 toxic chemical models and discusses five of the models in some depth. These are a steady state eutrophication model, a Lake Erie model, a physical-chemical model of toxic substances in lakes, a mass balance model of metals in lakes and the fugacity model mentioned earlier in this paper.

While many models assume a steady state equilibrium situation, this is not normally the case in nature because of localized emissions, often from multiple sources, changing meteorological, hydrological and other environmental conditions, and the nature and resistance of transport and transformation processes between compartments or phases such as air, water, sediment and biota (Clark et al. 1988). But models which attempt to incorporate all such factors will require a substantial allocation of resources for development, calibration and validation.

Output of Models

An example of the output of a modelling exercise might be summarized as follows: when discharged to water, the substance tends to partition between water, sediment, atmosphere and biota in the approximate proportions 20:20:59.9:0.1. The concentrations in sediment and biota can be as much as 1000 times that of the water column. The dominant degradation processes are atmospheric photolysis accounting for approximately 70%, and aquatic photolysis accounting for approximately 10%. The overall half-life of the substance in the system is 10 days.

The output from another model could generate a time series which might show a significant change in proportions until equilibrium is reached:

Compartment	48 Hours	25 Days
Water	48	0.8
Soil	25	0.5
Biota	0.8	<0.1
Air	3.8	11.4
Metabolized	2.9	11.0
Hydrolyzed	25	76.0

By undertaking sequential analyses on a number of chemicals, which would have similar use patterns and release situations, for example, PCB replacement compounds, a company or a regulatory agency might be able to eliminate some chemicals from their research program (Addison et al. 1983).

Limitations of Model

The constraints associated with use and choice of models have been described by many authors (IJC 1987):

- cost of use
- the assumptions may be inappropriate for the particular need or situation;
- documentation may not be available in usable form;
- the model may not include the important transport and transformation processes for the chemical and environment in question;
- the data requirements may be extensive or unrealistically attainable;
- the spatial or temporal resolution of the model may be inadequate;
- differences between predicted and actual concentrations of at least an order of magnitude can be expected;
- the model in question may have been inadequately calibrated and validated;
- the model may have designed for a limited range of situations;
- models generally do not provide averages and variances of concentrations.

As an example, the persistence screening model developed by Roberts et al. (1981) at the National Research Council of Canada is based on the following assumptions:

(1) that hydrodynamic processes do not result in net outflow or inflow to the system;
(2) that sedimentation is not a major removal process;
(3) that compartment size remains constant;
(4) that mixing is rapid relative to the rates of other processes;
(5) that pollutant concentration does not exceed its solubility in any compartment;
(6) that first order kinetics apply;
(7) that an equilibrium eventually describes the distribution of a chemical between compartments.

Some of these assumptions are common to many models but may not be appropriate in the situation or for the conditions being investigated. As indicated by Roberts et al. (1981), these assumptions bias the predictions toward worst case situations, in other words, they may overestimate the concentrations. This is not necessarily a problem if the model is being used for screening purposes, i.e. to determine what chemicals or groups of chemicals require more study, but that model is likely to be inappropriate for determining whether more treatment is actually required in a particular case.

Barnthouse and Van Winkle (1980) have recommended three guidelines that should be followed in choosing models:

1. The choice depends primarily on the goals of the program and on the quantity and quality of data available or obtainable.

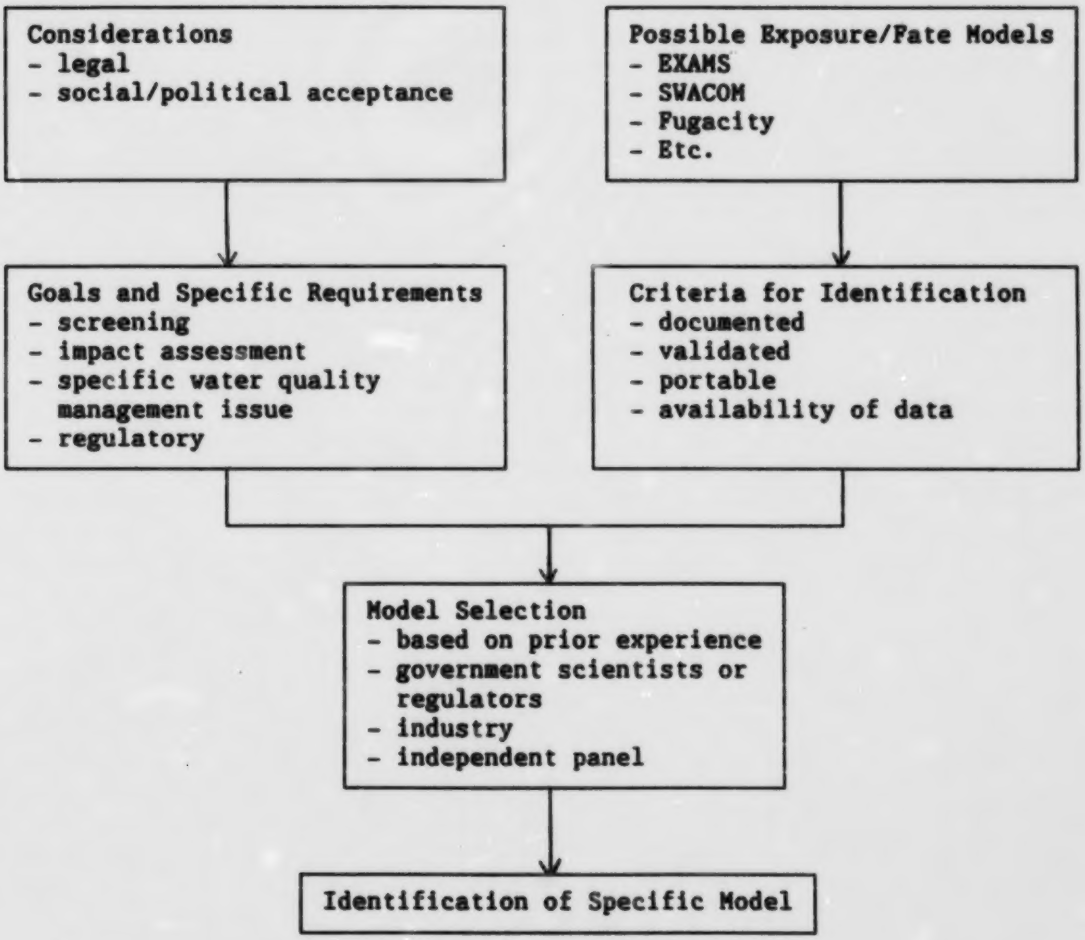
2. The model selected should be the simplest that is adequate to reduce possibilities of error.

3. The model should be chosen as early as possible in the program.

CONCLUSION

Models have a wide range of uses as Table 3 indicates. They are being utilised extensively in priority setting exercises in hazard assessment programs for example by the Organization for Economic Cooperation and

Figure 5 Scheme for selecting appropriate models
(after Barnthouse 1981)


Considerations
- legal
- social/political acceptance
Possible Exposure/Fate Models
- EXAMS
- SWACOM
- Fugacity
- Etc.
Goals and Specific Requirements
- screening
- impact assessment
- specific water quality management issue
- regulatory
Criteria for Identification
- documented
- validated
- portable
- availability of data
Model Selection
- based on prior experience
- government scientists or regulators
- industry
- independent panel
Identification of Specific Model

Development and the Ontario Ministry of the Environment. Similar hazard assessment can also be done at the watershed level to determine which pesticides might be of greatest concern. For example, Burridge and Haya (1989) undertook such an assessment of 68 pesticides used in Prince Edward Island. A similar approach could have been employed in an evaluation conducted by the Inland Waters Directorate of the potential of various pesticides to contaminate groundwater resources in the Maritimes. The initial screening involving soil leaching potential, use patterns and known human health concerns could be supplemented by one of the fate models to provide more detailed assessments (Gillis and Walker 1986).

As indicated earlier, exposure or fate models can also be utilized to consider the impact of different mitigative actions on concentrations in different media. For example, if chemical X was banned, the length of time required for the concentrations of that chemical to reach the water quality objective or standard could be calculated. The HSP-F model was designed to generate statistical information on the impacts and system responses of implementing basin-scale water quality management decisions.

The primary impetus for the development and application of exposure/fate models was the Toxic Substances Control Act of the United States. While some work has been done in Canada on exposure models (generic-fugacity; specific-Alberta pulp and paper mills) and more may be done as a result of the promulgation of the Canadian Environmental Protection Act, there are few if any requirements under legislation that would necessitate the use of exposure/fate models in environmental impact assessment, environmental protection or water quality management.

If such requirements are to be established there will have to be clear guidelines regarding the type of information that must be collected. This includes production and use patterns, emission rates, methods of release, the physical and chemical behaviour of the chemical in the environment (though benchmark chemicals and structure-activity relationships may provide some comparable data) chemodynamics, biotransformation and the identity of decomposition products.

Exposure and fate models clearly have applicability in environmental protection and water quality management. Care must be exercised however in selecting appropriate models (Figure 5). The assumptions on which the models were built, the degree of sensitivity analysis undertaken, and the extensiveness of the validation should be investigated. Finally, it remains to be reiterated, that models are tools to aid in decision making; they do not replace the experts nor the managers.

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Hazard Assessment II: The Role of Ecotoxicology in Assessing Aquatic Effects and Setting Water Quality Objectives

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INTRODUCTION

Canada is committed, most recently through the Canadian Environmental Protection Act (1988), to preparing formal hazard assessment reports on priority chemicals and substances known or suspected to be a threat to terrestrial, atmospheric and aquatic environments. Incorporated into such reports, as discussed recently at the CEPA Priority Substances Science Forum (Environment Canada/Health and Welfare Canada 1989), is scientific information on exposure (sources, environmental transport, transformation and levels, and population exposures) and toxicity (kinetics and metabolism, mammalian toxicology, effects on humans, and effects on various natural ecosystems).

The question of determining exposures, especially through the use of aquatic fate models, have been addressed by the previous speaker (Côté MS1989). This paper briefly describes how the toxic effects of substances on components of aquatic ecosystems are measured, and the role of such measurements in assessing hazard and setting water/sediment guidelines and objectives for ecosystem protection (Fig. 1). Only the role of eco-and aquatic toxicology for safeguarding aquatic ecosystems is described. It is assumed that the primary use of the aquatic environment addressed is the protection of aquatic life and aquatic ecosystems.

Therefore, following from Lloyd (1981) "water quality standards (i.e. guidelines and objectives) for AQUATIC LIFE PROTECTION are required only for those chemicals likely to occur in harmful concentrations within the aquatic environment. The first requirement is to define the extent to which the aquatic ecosystem is to be protected." Defining the level of protection requires a knowledge of safe and unsafe levels (i.e. threshold concentrations) for a range of biota, processes and habitats. It also requires a definition of "acceptable and unacceptable effects" (Stephan 1986). This paper briefly describes the measurement of acute and chronic effects to define levels of protection. Borgmann (MS1989) and MacDonald (MS1989) will address the status of field and ecological measures of ecotoxicology working towards the same goal.

HAZARD ASSESSMENT AND AQUATIC TOXICOLOGY - DEFINITIONS AND CONCEPTS

Certain definitions and concepts are essential to the process of deriving meaningful water quality guidelines and objectives.

Hazard assessment is "the prediction of the magnitude and duration of chemical concentrations in various segments of the environment, resulting from chemical use (and discharge), compared with the concentrations of the chemical in food, water or air which are known to be harmful to representative species, populations and ecosystems" (Kenaga 1979). Hazard assessment requires knowledge of the environmental fate of the chemical and knowledge of the toxicological properties of the chemical. Toxicity and hazard are not synonymous. Estimates of hazard are essentially ratios of expected exposure concentrations and effect concentrations. It should also be noted that "the shift from aquaria to microcosms to field studies is not concerned with toxicity; it is concerned with the real variables in hazard assessment, the exposure assessment". (Parrish et al. 1988). When completed, each hazard assessment leads to a risk assessment (involving hazard and the probability of the event) and ultimately to risk management (risk versus options to minimize it) of the specific problem.

One of the basic objectives of applied ecotoxicologists is "to place the right numbers and kinds of tests with ecologically significant test species into a workable systematic program to enable an early, cost-effective prediction of hazard at an acceptable level of certainty, to guide regulatory and industrial decision-making" (Maki 1983). This includes contributing to the establishment of water and sediment quality guidelines and objectives by applying appropriate threshold concentrations, if they exist, for recognized toxic substances, and estimating "safe" concentrations (Samoiloff and Wells 1984). Such estimates, which are the water and sediment guidelines and objectives, should be a major end result of all hazard and risk assessments.

Toxicology is the study of the adverse effects of chemicals in living organisms (Chiras 1985). Ecotoxicology is a branch of toxicology "concerned with the toxic effects of physical and chemical agents on living organisms, especially on populations and communities within defined ecosystems; it includes the transfer pathways of those agents and their interactions with the environment" (Butler 1978). The terms "acute, chronic, lethality and sublethality" often cause confusion. Acute means short-term, but in aquatic toxicology, acute often infers a short-term lethal test or response. Chronic is long-term or longer-term, and should be related to the generation time of the organism under study. Lethality is that toxicity which causes death by direct action. Sublethality is that toxicity which causes an effect at concentrations below those that directly cause death (Rand and Petrocelli 1985).

Some of the other terms/values crucial to the process of establishing water quality guidelines/objectives are: (from Rand and Petrocelli 1985)

Application Factor - a numerical, unitless value, calculated as the threshold chronically toxic concentration of a chemical divided by its acutely toxic concentration. It is generally calculated by dividing the limits on the maximum acceptable toxicant concentration (MATC) by the time-independent LC50 or 2- or 4-day LC50's. Empirically-derived application factors can be used for setting objectives.

Incipient LC50 - the concentration of a chemical which is lethal to 50 % of the test organisms as a result of exposure for periods sufficiently long that acute lethal action has essentially ceased. It is the concentration below which mortality is not expected, upon indefinite exposure, unless a sublethal effect becomes lethal.

MATC or maximum acceptable toxicant concentration - the hypothetical toxic threshold concentration lying in a range bounded at the lower end by the highest tested concentration having no observed effect (NOEC) and at the higher end by the lowest tested concentration having a significant toxic effect (LOEC) in a life cycle (full chronic) or partial life cycle (partial chronic) test.

ROLE OF TOXICOLOGICAL ASSESSMENTS

During the past three decades, a great deal of effort has been devoted to the development, improvement, standardization and application of laboratory toxicity assays. They have been used as both research and regulatory tools - identifying, assessing and controlling problems, and monitoring to determine compliance with limits and standards, the effectiveness of remedial actions, and the occurrence of unpredicted or unexpected threats.

According to Hunn (1989), recent work, especially in North America, has emphasized:

- standardization of test procedures in aquatic toxicity testing;
- definition of biotic and abiotic factors influencing the measures of aquatic toxicity;
- broader application of aquatic tests. Examples include:
    - estimation of half-life of biological activity;
    - on-site toxicity assessment;
    - acute toxicity of sediments;
    - toxicity of chemical mixtures;
    - development of quantitative - structure - activity relationships (QSARs);
    - registration of chemicals, such as pesticides and piscicides;
    - use in ecological hazard assessment procedures.

Following from work on the Environment Canada "Strategic Framework for Toxic Chemicals Environmental Sensing" (Eisenhauer pers. comm.; Wells and Côté 1988), Sergy (1987) described the roles and applications of aquatic biological tests for environmental protection as being:

A. PROBLEM IDENTIFICATION (1. Identify and screen potential hazards; 2. Signal changes in environmental quality);

B. PROBLEM ASSESSMENT (3. Assess hazards; 4. Assess receiving environment effects; 5. Set ambient environmental quality objectives);

C. **CONTROL OR INTERVENTION** (6. Evaluate waste treatment and disposal methods; 7. Apply site-specific controls; 8. Apply a national level of protection; 9. Enforce regulatory standards);

D. CONTROL EVALUATION (10. Monitor compliance-to-control measures; 11. Monitor effectiveness of control measures; 12. Monitor for long-term environmental trends).

At this workshop, we are most concerned about problem assessment and the setting of appropriate environmental quality objectives.

Following from Sergy (1987), there is now a recommended core set of aquatic testing procedures for Environment Canada, which are used in various hazard assessments. Formal protocols for each test are in preparation.

Most recently, in an important new synthesis (Levin et al. 1989), Kelly and Harwell (1989) have focused on "how to characterize an ecosystem's response to disturbance and its subsequent recovery upon removal of stress". (Table 1). Knowledge of response-recovery processes and rates, after exposure to conservative and non-conservative materials and chemicals, is vitally important to the setting of realistic water quality objectives.

Table 1. Exposure, response, and recovery characterization, and uncertainties. (Kelly and Harwell 1989)

EXPOSURE OF ECOSYSTEMS TO ANTHROPOGENIC STRESS
- *Fate, transport,* and environmental *modification*
- *Duration, frequency, intensity,* and *novelty* of *exposure*
- *Differential exposure regime* within an ecosystem

RESPONSE OF ECOSYSTEMS TO ANTHROPOGENIC STRESS
- Effects on *components* of ecosystems
- Effects on *processes* of ecosystems
- *Relevant scales* and *characterization* of response
- *Endpoints* for response characterization
- *Relevant indicators* for endpoints

RECOVERY OF ECOSYSTEMS FROM ANTHROPOGENIC STRESS
- Indicators for *components* and *processes*
- *Irreparable harm* and (or) the *ability to adapt*
- *Resilience* and *homeorhesis*
- *Scales of physical* and *biotic renewal processes*

UNCERTAINTIES
- *Variability in exposure*
- *Variability in ecosystems*
- *Extrapolation across types of stresses*
- *Extrapolation across types of ecosystems*

ACUTE TESTING OF CONTAMINANTS

Several points are worth making about the roles and methodologies of acute tests on chemicals.

Short-term lethal and sublethal toxicity tests can be used to answer many questions (modified from Buikema et al. 1982; MacGregor and Wells MS1984; Wells 1988):

1) At what concentrations is the material acutely toxic? Is it considered to have low, moderate or high acute toxicity?

2) Are acute sublethal effects occurring? What type are they? At what concentrations? What is their dependency on the life cycle of the exposed species? Can the organisms recover from these effects?

3) Which waste (or waste component) is most toxic? Can the source of toxicity be identified, through a system such as the Toxicity Identification Evaluation of EPA?

4) Which organisms (or group of organisms) are most sensitive? Is the comparative toxicology well understood for the contaminants of concern?

5) Under that conditions are the wastes most toxic?

6) Does toxicity change when the material goes into the environment? If so, can the reason(s) for the change be readily identified?

7) How much of the receiving environment is affected? For how long? What is the potential for delayed effects, or long-term exposures?

8) What are the short term effects of episodic spills or events?

The design, conduct and analysis of acute tests have been described in many places (Persoone et al. 1984; APHA 1985; Rand and Petrocelli 1985; Buikema et al. 1982; Munawar et al. 1989).

It is important to recognize that a number of test methods are gaining precedence. The most highly useful tests, according to Buikema et al. (1982) were acute lethality, embryo-larval, and reproductive impairment. To this could be added the criteria of small size (microscale tests, see Blaise et al. 1988); sensitivity to toxicants; well-defined and biologically important endpoints; and inexpensive, rapid and logistically simple methods (Hansen 1986 (?); Connell 1987; Day et al. 1989; Goldberg and Frazier 1989). A whole field for developing such acute tests has developed (Dutka, pers. comm.). Manuals of methods for the most commonly used tests abound (eg. Peltier and Weber 1985).

The choice of test organisms and toxicological responses depends in part upon the objective of the study. The choice of species and the range of responses become more important when moving from simply comparing relative toxicities of substances to conducting impact assessments based upon site-specific information.

Many factors influence the acute toxicities of chemicals. They include physico-chemical (chemical composition, behaviour in water) and biological (phylogeny, life cycle stage, physiology). For the purposes of water or sediment quality guidelines, these factors need to be well-understood, and their study represents a large part of current effort in the ecotoxicological field. The recent paper by Mayer and Ellersieck (1988) is worth noting, as it deals with factors influencing the toxicity of chemicals, largely pesticides; temperature was a primary factor.

As stated above, two very active areas in acute aquatic toxicology are the preparation of standard methods for a wide range of testing approaches, and the development and refinement of applied micro-scale tests and their incorporation into hazard assessment schemes. Such acute tests are being used in support of water quality guidelines and objectives. Stephan (1986) stated that "a major goal of applied aquatic toxicology is to make a useful prediction concerning whether or not a specific addition of a toxic agent to a particular aquatic ecosystem will cause any unacceptable effect on that ecosystem". Acute tests, both lethal and sublethal, produce data used for establishing acceptable ("Safe") and unacceptable ("Unsafe") levels of substances upon which water quality guidelines/objectives are based. Acute tests also contribute to defining what is meant by "unacceptable effects on aquatic ecosystems", upon which the processes of hazard assessment and deriving water quality criteria (i.e. guidelines/objectives) are wholly dependent (Stephan 1986). More will be said on this point below.

CHRONIC TESTING OF CONTAMINANTS

Most effort in environmental and aquatic toxicology in recent years has been in studying and understanding the chronic, sublethal effects of contaminants on a wide range of species and biological processes, and estimating the concentrations causing those effects. The derivation of sublethal response thresholds, shown graphically by Waldichuk (1985) (Fig. 2), has been one goal of this work. Estimations are made of "safe" concentrations, or MATCs (maximum acceptable toxic concentrations). These contribute directly to the process of deriving water quality guidelines and objectives.

Once contaminants are bioavailable and bioaccumulated (i.e. exposure has occurred, and materials have been taken up by biota), sublethal effects may occur at a range of levels of biological organization (Fig. 3) (Widdows 1983; Sheehan _et_ _al_. 1984; Capuzzo and Kester 1987): subcellular (cytochemical and biochemical responses); cellular/tissue (cellular and tissue responses); organismic (physiological responses); population (changes in population structure and function); and community (changes in community structure and function). At the level of the individual, these can be categorized as behavioural, biochemical, physiological, morphological/pathological, and altered performance (Sheehan _et_ _al_. 1984). Examples of such effects at the individual level abound (eg. behavioural, morphology and pathology, reproduction).

Capuzzo (1981) (Table 2) summarized general sublethal responses to pollutant stress, emphasizing the adaptive and destructive responses at particular levels of biological organization.

Table 2. Summary of responses to pollutant stress. (Capuzzo 1981)

Level	Adaptive response	Destructive response	Result at next level
Biochemical-cellular	Detoxification		Adaptation of organism
		Membrane disruption Energy imbalance	Reduction in condition of organism
Organismal	Disease defense Adjustment in rate functions Avoidance		Regulation and adaptation of populations
		Metabolic changes Behaviour aberrations Increased incidence of disease Reduction in growth and reproduction rates	Reduction in performance of populations
Population	Adaptation of organism to stress No change in population dynamics		No change at community level
		Changes in population dynamics	Effects on coexisting organisms and communities
Community	Adaptation of populations to stress		No change in community diversity or stability Ecosystem adaptation
		Changes in species composition and diversity	Deterioration of community
		Reduction in energy flow	Change in ecosystem structure and function

When chronic sublethal experiments are well designed, they can produce reliable estimates of MATCs (eg. such as from chronic tests with fathead minnow eggs and larvae).

A number of sublethal biological responses are being considered for monitoring (eg. marine biomonitoring, Waldichuk 1985). At the present time, based on two marine experimental workshops, a number of sublethal tests and response variables are showing greatest promise: biochemical (microsomal xenobiotic metabolizing systems; metallothionein induction); tissue (histopathology); physiology (scope-for-growth; viability of molluscan embryos; lipid composition); and community (multivariate analysis of benthic community structure) (See Bayne et al. 1988).

SPECIAL TOPICS

A number of topics require special mention, as they are of particular importance in evaluating the biological effects of environmental contamination.

These include: the comparison of laboratory measured effects with those anticipated or measured under natural conditions (i.e. laboratory-field comparisons, multi-species toxicity testing); the effects of modifying factors on the results of toxicity tests; the ecotoxicology of contaminated sediments; the assessment of the joint or combined toxicity of chemicals; and the choice of biological variables to measure in effects-monitoring programs.

Each of these topics is an active research field at the present time, and each is and will contribute new principles and concepts important to accurately establishing guidelines and objectives for toxic substances and other variables. As an example, Lee Harding (MS1989) addresses current approaches to sediment quality objectives.

SOME LIMITATIONS OF ECOTOXICOLOGICAL ASSESSMENTS IN THE LABORATORY

As recently stated by Peakall (1989), "ther are vast numbers of protocols for hazard assessment, developed for the EEC, EPA, OECD". The ASTM also is particularly busy in this area. But there are limitations to the approaches, the main ones being that they deal with the testing of single chemicals and they consider testing on a single species or on simple assemblies of species.

Other limitations abound and should be kept in mind. Protocols are only considering a few species and biological processes. Our knowledge of extrapolation from one species to another (i.e. comparative ecotoxicology) is very limited. There is limited knowledge of the effects of metabolites and other environmentally transformed products of the parent chemicals. Relatively little of the chronic sublethal toxicology has led to estimates of threshold concentrations (i.e. MATCs and NOELs). The protocols do not take into account cumulative effects of chemicals, or compensatory responses (such as acclimation) of organisms. However, these problems are well recognized and are stimulating the next generation of basic and applied ecotoxicological research.

SOME CURRENT APPROACHES TO HAZARD ASSESSMENT

A number of countries have developed or are developing hazard assessment schemes for chemicals, effluents and environmental samples.

For example, Canada has the Quebec Hazard Assessment Scheme (HAS) of Blaise et al. (1988) which is an integrated ecotoxicological approach for hazard assessment of industrial effluents. There is the Great Lakes Approach (Strachan 1988) for the assessment and control of chemicals entering the Great Lakes Basin. In the mid-1980s, Environment Canada also developed an environmental sensing framework for toxic chemicals which incorporated direct linkages between hazard assessments, environmental quality objective setting, and management decision-making (Eisenhauer, pers. comm., Wells and Côté 1988).

Internationally, Sweden has the ESTHER approach (Systems for Testing and Hazard Evaluation of Chemicals in the Aquatic Environment) (Ladner 1988), which has procedures for initial and advanced hazard assessment (Table 3). The USA has many schemes, particularly stimulated by the ASTM, EPA, TOSCA and SETAC, and workers such as John Cairns and Alan Maki (Cairns et al. 1978; Maki and Bishop 1985; Bergman et al. 1986). West Germany has formal schemes that link hazard assessment directly to the processes of deriving environmental quality standards, criteria and objectives (Hansen, pers. comm.) (Fig. 1).

All of these hazard assessment schemes have a toxicological/ecotoxicological component, and all are supportive of the development of practical, small scale, predictive toxicity methodologies.

CONTRIBUTIONS OF TOXICOLOGY TO WATER AND SEDIMENT QUALITY GUIDELINES AND OBJECTIVES

Aquatic toxicology has been contributing to the setting of guidelines and objectives for water since the 1950s. Products of this work are the well-known "blue", "red" and "gold" books (see, for example, EPA 1986), the periodic Federal Register publications of freshwater and marine "criteria" (eg. Federal Register 1979), and Canada's water quality guidelines (CCREM 1987).

Lately, the emphasis has been on sediment quality criteria (guidelines) (Chapman 1986; Chapman et al. 1987; many papers at SETAC and other meetings). Of all of the areas currently showing how toxicology is contributing to hazard assessment, this one is drawing the most scientific and managerial attention. Finally, sediments are being recognized as needing protection due to being temporary or permanent sinks for many persistent substances; sediment questions are challenging the toxicologists and ensuring continued input of state-of-the-art ecotoxicology to guidelines/objectives development and use.

Table 3. The two facets or phases of environmental hazard assessment of chemicals in the ESTHER approach. The initial phase uses a set of simple, rapid and relatively inexpensive methods to obtain a first, preliminary idea about the potential hazard of the chemical under assessment. For chemicals of concern, a more advanced approach is then applied. (Ladner 1988)

INITIAL HAZARD ASSESSMENT	ADVANCED HAZARD ASSESSMENT
Exposure analysis	
Degradation studies in complex systems (black-box type of approach) - mixed-culture biodegradation test; +/-type of answer	Transformation and degradation studies mechanism-oriented; identification of metabolites and their properties - pure cultures of bacteria, isolated from environmental samples - delineation of separate steps in biotransformation of xenobiotics
Estimating bioaccumulation potential from physico-chemical properties - octanol/water partitioning - molecular mass	*In vivo* bioaccumulation studies: uptake (penetration of biological membranes), distribution, metabolism, excretion (analysis of conjugates)
Estimating environmental distribution through theoretical fugacity calculations	Studies of sorption to sediment; confirmation of results from laboratory fate analysis in controlled mesocosm studies
Effect analysis	
Single-species, short-term tests with hardy test organisms under standardized laboratory conditions; Clearcut, qualitative endpoints: - mortality - immobilization - growth/no growth Sub-acute studies with isolated species	Holistic approach Multispecies experimental systems Structure and function of communities Guild analysis Long-term exposure of natural associations of organisms in microcosms, mesocosms or limnocorrals Identification of most sensitive components in ecosystems or of key-stone species Description of indirect effects, changes in nutrients and energy cycling

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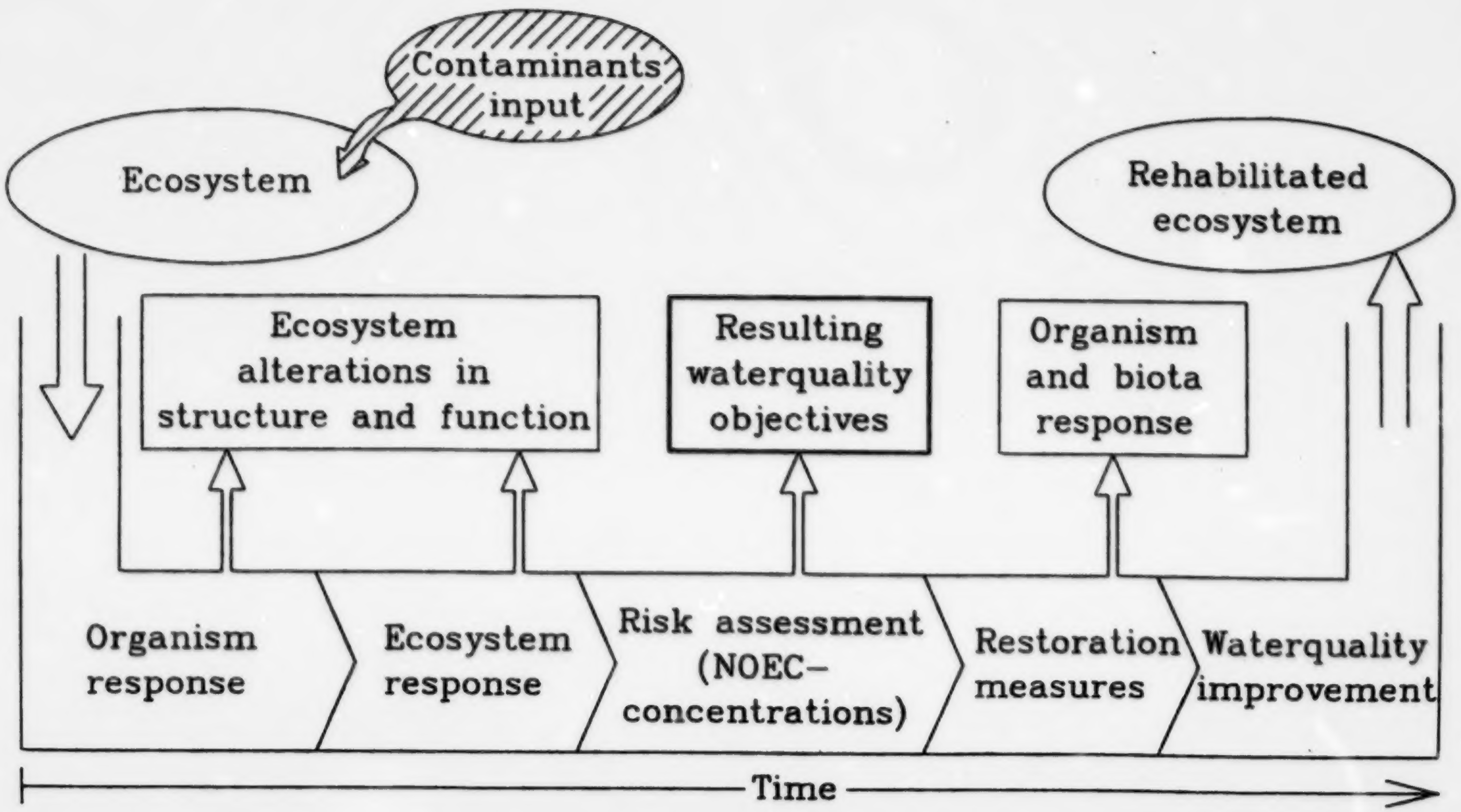
Contaminants input
Ecosystem
Rehabilitated ecosystem
Ecosystem alterations in structure and function
Resulting waterquality objectives
Organism and biota response
Organism response
Ecosystem response
Risk assessment (NOEC–concentrations)
Restoration measures
Waterquality improvement
Time

Figure 1. **Hazard Assessment Scheme (Hansen, pers. comm.)**

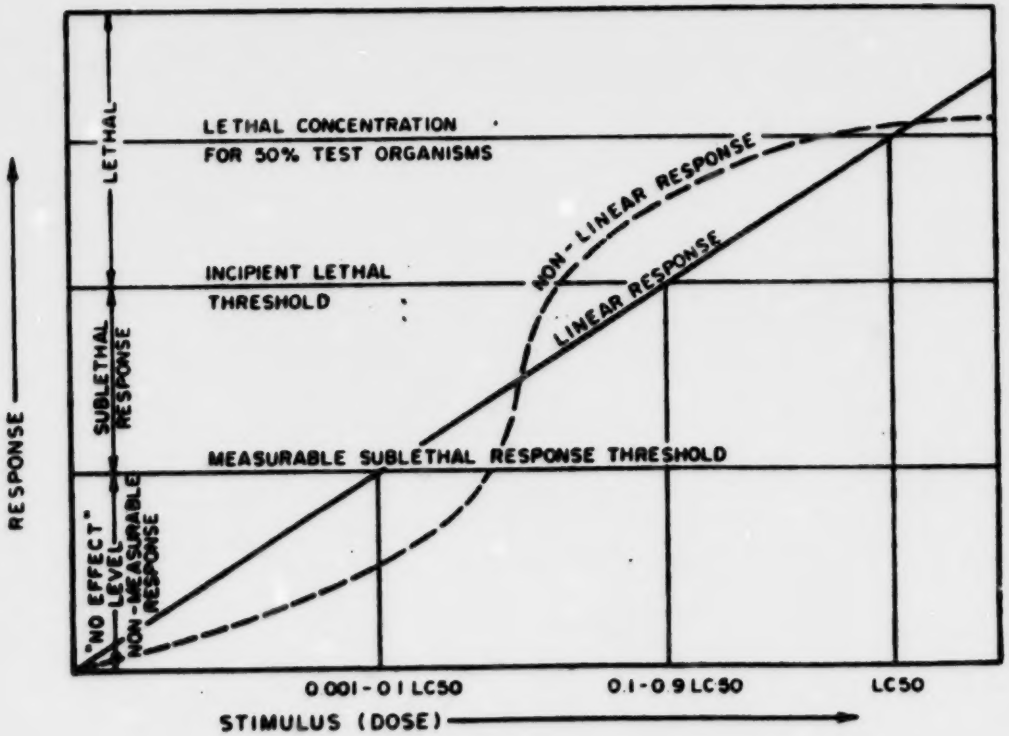

Figure 2. Hypothetical relationship of the response of an aquatic organism to concentration of a chemical, showing some significant points and regions on the curve. (Waldichuk 1985)

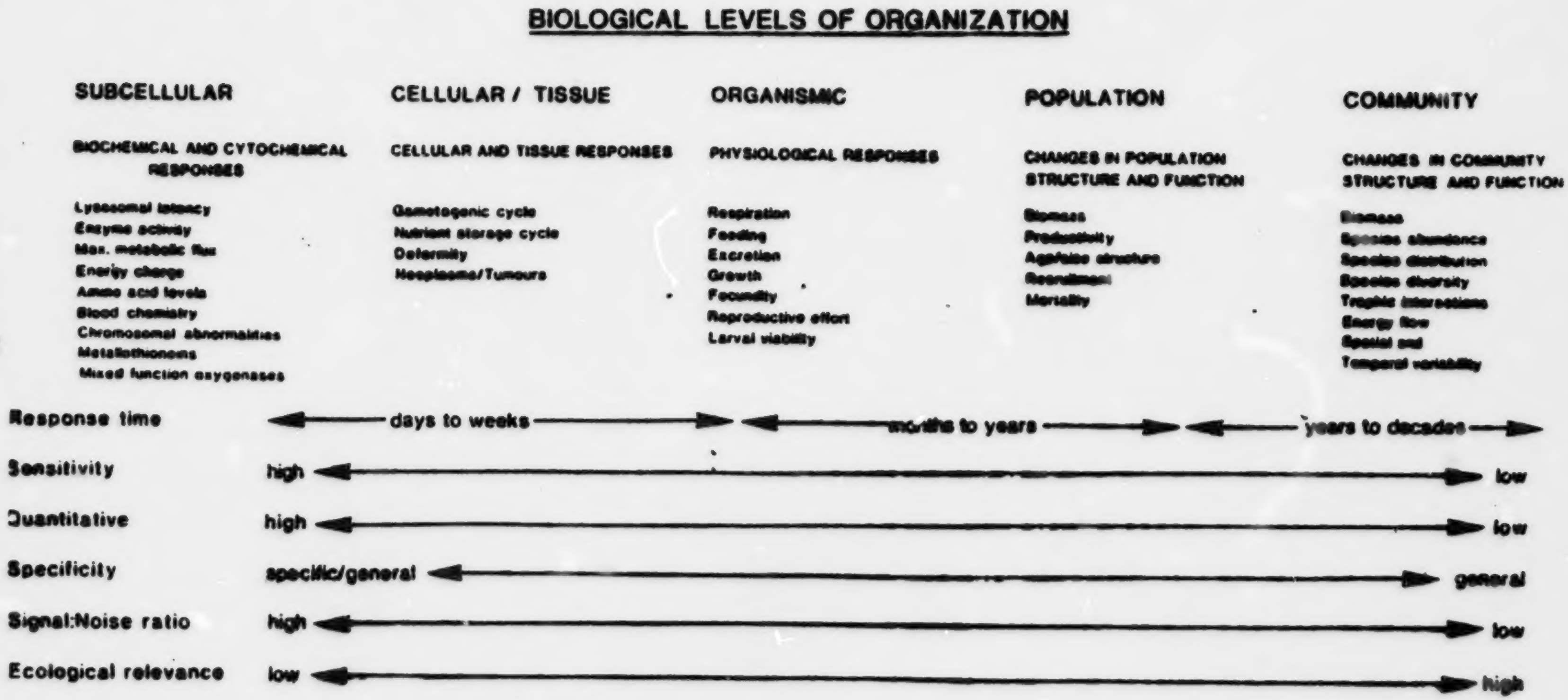

Figure 3. Biological responses suitable for measuring the effects of pollution at five levels of biological organization (Widdows 1983).

Using *in situ* Bioassays as a Basis for the Development of Water Quality Objectives: A Case Study of the Arkansas River.

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ABSTRACT

The Canadian Council of Resource and Environment Ministers (CCREM, 1987) developed and published the Canadian Water Quality Guidelines to provide information to aquatic resource managers, industry, and the public. These guidelines recommend characteristics of aquatic ecosystems that are required to support and maintain designated water uses. While these national guidelines provide an excellent source of information on use requirements, there are many situations that require additional data to make rational decisions on how aquatic resources should be used. In these cases, site-specific water quality objectives are needed. This paper describes an approach to using *in situ* bioassays as a basis for the development of water quality objectives. The results of tests run in the Arkansas River basin are reported to illustrate the value of these methods.

INTRODUCTION

The residents of southeastern Colorado are dependent on the Upper Arkansas River and its tributaries to support a variety of water uses. A number of communities rely on the waters of the basin to provide domestic freshwater supplies. Water is also drawn from the system to sustain a number of agricultural activities, including irrigation and livestock watering. In addition to the consumptive uses, the Arkansas River supports a diversity of aquatic life, historically (up to 1984) including a gold medal brown trout fishery. Recreational water uses, such as swimming, canoeing, and viewing are also important in certain parts of the basin.

However, many of these designated water uses are currently in jeopardy due to historical and/or ongoing developmental activities within the basin. Existing and abandoned mining operations and other developments in the basin have resulted in unfavourable water quality conditions in affected areas. Currently, direct effluent discharges from active and inactive mines (primarily via the Leadville Mine Drainage Tunnel and California Gulch), uncontrolled drainage from abandoned mine sites (largely affecting California Gulch), and desorption from metal-contaminated stream-bed sediments (downstream of California Gulch) represent major sources of toxic metals to the stream system. Loadings of other contaminants to the river result from municipal discharges and diffuse non-point sources (Figure 1).

The available water quality data clearly demonstrate severe heavy metals contamination in the Upper Arkansas River from the Leadville Mine Drainage Tunnel seepage area to Buena Vista, Colorado. Fisheries studies undertaken by Colorado Department of Wildlife (Nehring 1986) indicated that both the diversity and abundance of aquatic biota within the areas affected by metals pollution had been markedly reduced relative to control sites. Of particular concern with respect to the maintenance of a high quality fishery was the poor survival of brown trout after attaining sexual maturity.

Preservation and restoration of designated water uses in the Upper Arkansas River, as mandated under the Clean Water Act, requires implementation of a water management strategy specific to the river basin. The components of this strategy include: designation of water uses, description of existing water quality and quantity conditions, development of water quality objectives, and formulation of waste treatment strategies which will lead to improvements in water quality. Further, water quality and biological monitoring is required to correlate enhanced waste treatment with changes in water quality and restoration of water uses, especially aquatic life.

This study was designed to assess the ambient water quality conditions in the Arkansas River and its tributaries from above Leadville to below Buena Vista, Colorado. A biomonitoring approach, using Ceriodaphnia dubia and fathead minnow (Pimephales promelas) bioassays, was selected to provide information directly applicable to the water quality objectives development process (Environmental Protection Agency 1983, Stephan et al. 1985). Subsequent comparison of bioassay results with instream biological monitoring data provided a basis for assessing the applicability of this approach for defining conditions conducive to the rehabilitation of the brown trout fishery.

MATERIALS AND METHODS

In the Arkansas River basin, in situ bioassays consisted of acute and short-term chronic toxicity tests (Environmental Protection Agency 1985a & b). The acute toxicity of water from Arkansas River basin sites to Ceriodaphnia dubia and fathead minnows was determined in 48 and 96 hour static renewal tests, respectively. The short-term chronic toxicity tests consisted of 7 day static renewal tests designed to measure the chronic survival and reproduction of Ceriodaphnia dubia and the chronic survival and growth of fathead minnows.

Toxicity profiles were developed for 18 sites in the Arkansas River basin using unspiked water samples in the tests. Where site water resulted in acutely toxic conditions to greater than 50% of the test organisms, a dilution series was set-up to facilitate determination of the lethal concentration (LC_{50}). Comparison of the results of these short-term tests provided a means of assessing the relative toxicity of water from each site to the test organisms.

RESULTS

1. Acute Toxicity Tests

A listing and description of twelve key sampling sites within the Arkansas River basin is presented in Table 1 (Figure 1). The results of the in situ acute toxicity bioassays conducted on Ceriodaphnia dubia and fathead minnows in the Arkansas River basin during September, 1987 are presented in Table 2. These results indicated that, of the two species, C. dubia was a more sensitive indicator of heavy metals contamination. Further, comparison of the C. dubia toxicity data with brown trout population data suggested that this crustacean was a good indicator of metal toxicity on instream biota. High levels of toxic metals (Figure 2) were clearly responsible for the low survival of test organisms downstream of the Leadville Mine Drainage Tunnel and California Gulch. Lethal concentrations (LC_{50}) of California Gulch water to fathead minnows were in the order of 1 - 2%, depending on duration of exposure and the source of the parent water used for sample dilution (Table 3).

More recent data (Table 4), collected by the United States Geological Survey (unpublished data) in May, 1989, suggest significant temporal variability in water quality conditions. These results indicate that Tennessee Creek is contaminated with metals originating from St. Kevins Gulch during high flow conditions. In addition, acutely toxic conditions persisted farther downstream than during the autumn sampling period. The information generated on the bioavailability of metals associated with sediments (as determined using filtered and unfiltered water samples) was inconclusive.

A brief overview of water quality data collected during the course of the bioassays is provided in Table 5, and presented graphically in Figure 3. Comparison of instream levels of toxic metals (adjusted to a standard water hardness; 50 mg/L) with final acute values reported in the Environmental Protection Agency (1985c & d, 1987) water quality criteria documents (for hardness = 50 mg/L) suggests that zinc and, to a lesser extent, cadmium are the elements primarily responsible for the observed acute toxicity.

2. Short-term Chronic Survival, Reproduction and Growth Tests

The results of the short-term chronic survival tests conducted on C. dubia and fathead minnows are reported in Table 2. Survival rates of fathead minnows were similar for both acute (2- or 4-days) and short-term chronic (7-days) exposures, indicating acute:short-term chronic ratios of close to one. For C. dubia, acute:short-term chronic ratios were higher. The C. dubia reproduction and growth tests provided little additional insight into the biological significance of ambient water quality conditions in the Upper Arkansas River. This is not particularly surprising considering that water from most of the sampling sites was acutely toxic to this species.

DISCUSSION

In situ bioassays are, increasingly, gaining recognition as useful and important tools in water management. In the United States, bioassays form an integral part of the water quality standards development process. With the methodology currently available, acute and short-term chronic information can be generated in a reasonable time-frame and on a cost-effective basis. By using standard test species, the results of in situ bioassays can be compared to those obtained at different locations and at different times. These tests, therefore, provide detailed information on the relative toxicity of a water source at a specific location.

Comparison of the results of these short-term tests with data collected on resident brown trout populations in the Upper Arkansas River emphasizes the utility of this approach. Annual evaluations conducted since 1981 (Nehring and Anderson 1981, 1982, & 1985; Nehring et al. 1984) indicate rapid growth and good survival of brown trout during the first two years of their life. However, poor survival of sexually mature brown trout (age 3+ and older) has also been documented. Nehring (1986) suggested that the cumulative effects of siltation, poor food supplies, heavy metal pollution, and the stresses associated with spawning activities (ie. abatement of feeding, poor forage base, and depletion of fat and other energy reserves) resulted in high mortality rates among the sexually mature fish. While the assessments of the effects on primary and secondary productivity are not yet available, it is clear that elevated levels of heavy metals have translated into reduced populations of brown trout (Table 2).

Development of site-specific water quality objectives applicable to the restoration of the Upper Arkansas River requires integration of diverse information relating to aquatic environmental quality. The situation in the Upper Arkansas River basin is particularly complex because aquatic biota are simultaneously exposed to a number of contaminant challenges. The combination of contaminants (zinc, copper, cadmium, suspended sediments, etc.) may result in toxic effects that are not immediately predictable using the literature based toxicity models. Therefore, water quality objectives that recommend numerical values for individual contaminants alone may be under- or over-protective of instream water uses.

The ultimate authorities on the quality of water are the organisms that are exposed to it, not the lawyers, engineers, and environmental scientists who litigate and study it. Data obtained from studying these living models, if properly generated and interpreted, provide empirical evidence with which to assess the quality of the aquatic environment. Further, correlation of biotic responses with other environmental variables, such as habitat, flow, water chemistry, and pollution discharges, provides a basis for water management decision making.

The existing water quality in the Arkansas River is such that acutely lethal conditions exist from the Leadville Mine Drainage Tunnel area to Buena Vista (approximately 50 river miles), based on short-term Ceriodaphnia toxicity data. An immediate goal in the clean-up of the Arkansas River is the elimination of acutely toxic conditions in the basin. In this situation, narrative water quality objectives appear to be most appropriate. These interim water quality objectives (1992-1995) state that there should be no acute toxicity to Ceriodaphnia anywhere in the Upper Arkansas River. Elimination of acute toxicity of waters originating from the Leadville Mine Drainage Tunnel, St. Kevins Gulch, and California Gulch should result in favourable water quality conditions for brown trout and other aquatic biota throughout most of the river basin. These interim objectives will require review and revision as the clean-up progresses to provide for the elimination of chronic toxicity, adverse reproductive effects, and/or harmful levels of bioaccumulation of metals in stream biota, if these are identified as ongoing problems.

Dose (as measured by concentrations of copper, cadmium, and zinc)/ response (mortality of indicator species) relationships generated for various stream reaches provide a means of defining guideline levels of metals in water. These supplementary objectives, in conjunction with appropriate dilution models, provide a basis for developing strategies for mitigation of metals toxicity problems in the river basin.

CONCLUSIONS

The Upper Arkansas River is currently being impacted through the degradation of water quality due to a number of anthropogenic activities. Heavy metals pollution originating from operating mines, abandoned mine sites, and contaminated stream-bed sediments has resulted in elevated levels of copper, cadmium, and zinc in the basin. Impacts on biological resources include depressed primary and secondary productivity, and severe reductions of brown trout populations. The long-term goals (1995 - 2000) for the basin include improvement of the instream water quality conditions and re-establishment of a high quality brown trout fishery. Achievement of these water management goals requires abatement of heavy metals contamination.

In the Arkansas River basin, *in situ* bioassays provided a scientifically defensible basis for the development of site-specific water quality objectives. Site-specific water quality objectives for the Arkansas River will contribute to the improvement of water quality by providing guidance to water managers on which sites and variables to focus their clean-up efforts. In addition, these same objectives will provide a yardstick against which the success of clean-up efforts may be measured.

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Table 1. Description of sampling sites in the Arkansas River basin.

Sample Site	Description
EF-1	East Fork of the Arkansas River approximately 12 miles upstream of the confluence with Tennessee Creek and downstream of the molybdenum mine. This control site corresponds to river mile 2.0.
EF-2	East Fork of the Arkansas River approximately 4 miles upstream of the confluence with Tennessee Creek. This control site corresponds to river mile 10.0.
EF-6	East Fork of the Arkansas River immediately upstream of the confluence with Tennessee Creek and downstream of the Leadville Mine Drainage Tunnel. Corresponds to river mile 14.0.
TC-1	Tennessee Creek near mouth (downstream of St. Kevins Gulch).
AR-1	Arkansas River downstream of the confluence of the East Fork of the Arkansas River and Tennessee Creek. Corresponds to river mile 14.1.
AR-2	Arkansas River downstream of the confluence of the East Fork of the Arkansas River and Tennessee Creek. Corresponds to river mile 16.8.
AR-3	Arkansas River downstream of California Gulch. Corresponds to river mile 16.9.
AR-4	Arkansas River upstream of Iowa Gulch. Corresponds to river mile 19.0.
AR-5	Arkansas River downstream of Iowa Gulch. Corresponds to river mile 21.5.
AR-7	Arkansas River upstream of Lake Creek. Corresponds to river mile 29.0.
AR-9	Arkansas River downstream Buena Vista. Corresponds to river mile 53.2.
AR-10	Arkansas River downstream of Chalk Creek. Corresponds to river mile 60.6.

Table 2. Toxicity profile[1] for selected sampling sites in the Arkansas River basin (Sept. 22-30, 1987).

Sample Site	Percent Survival						Brown Trout Population (fish/ha)[2]
	Ceriodaphnia dubia			Fathead Minnow			
	24hr	96hr	168hr	24hr	96hr	168hr	
EF-1	100	100	100	95	95	90	
EF-2	100	100	100	100	100	100	4407
EF-6	0	0	0	95	95	95	
AR-1	80	10	10	95	95	95	
AR-2	100	40	20	100	100	100	421
AR-3	0	0	0	5	0	0	est. 0
AR-4	10	0	0	85	85	85	
AR-5	90	20	20	100	85	85	
AR-7	100	70	60	100	95	95	279
AR-9	100	100	60	100	95	95	320
AR-10	100	100	100	100	100	100	

1. Source: USEPA unpublished data.
2. Data reported in Nehring (1986) was collected from August, 1977 to April, 1980.

Table 3. Toxicity[1] of water originating from California Gulch.

Dilution Water	Ceriodaphnia dubia[2] Exposure(hr)	Ceriodaphnia dubia[2] LC_{50}	Fathead Minnow Exposure(hr)	Fathead Minnow LC_{50}
EF-2	24	N/A	24	1.89
	96	N/A	96	1.55
	168	N/A	168	1.50
AR-2	24	N/A	24	1.06
	96	N/A	96	0.95
	168	N/A	168	0.95

1. Toxicity expressed as percent California Gulch water.
2. N/A = Not available.

Table 4. Toxicity profile for selected sampling sites in the Arkansas River basin (May 22-30, 1989).

Sample Site	Percent Survival[1] Ceriodaphnia dubia Unf.[2]	0.1um[2]	0.01um[2]	Fathead Minnow Unf.[2]	0.1um[2]	0.01um[2]
EF-1	95	50		100	100	
EF-6	0	10		95	95	
TC-1	60	15		40	10	
AR-1	35	60		95	95	
AR-2	80	95		40	45	
AR-3	0	0	0	5	0	0
AR-7	0	0	65	35	0	20
AR-10	90	100		55	45	

1. Exposure period was 96 hours.
2. Unf. = unfiltered sample, 0.1um = sample filtered through a 0.1 um filter, 0.01um = sample filtered through a 0.01 um filter (source: USGS unpublished data).

Table 5. Summary of water quality data for selected water quality variables and sampling sites in the Arkansas River basin (Sept. 22-30, 1987)[1].

Sample Site	Alk (mg/L)	Hard (mg/L)	pH	Cd (ug/L)	Cu (ug/L)	Zn (ug/L)
EF-1	60.7	52.9	8.1	1.01	2.02	56.0
EF-2	129	99.5	8.4	0.47	1.07	8.4
EF-6	138	161	8.5	0.44	0.74	144
AR-1	116	132	8.4	0.44	0.89	124
AR-2	123	127	8.3	0.39	0.97	90.6
AR-3	119	173	8.1	3.50	7.37	1100
AR-4	80.7	88.0	8.6	1.26	3.88	201
AR-5	90.0	103	8.7	0.48	1.09	241
AR-7	67.4	68.2	8.9	0.78	2.17	67.7
AR-9	74.5	70.2	8.1	0.78	1.74	81.8
AR-10	101	80.7	8.4	3.97	1.49	32.4

1. Source: USEPA unpublished data. Each value represents the mean of at least seven individual measurements. Metals concentrations were adjusted to effective concentrations at hardness = 50 mg/L to permit assessment to toxic equivalents (see text for explanation).

Figure 1. Upper Arkansas River Basin.

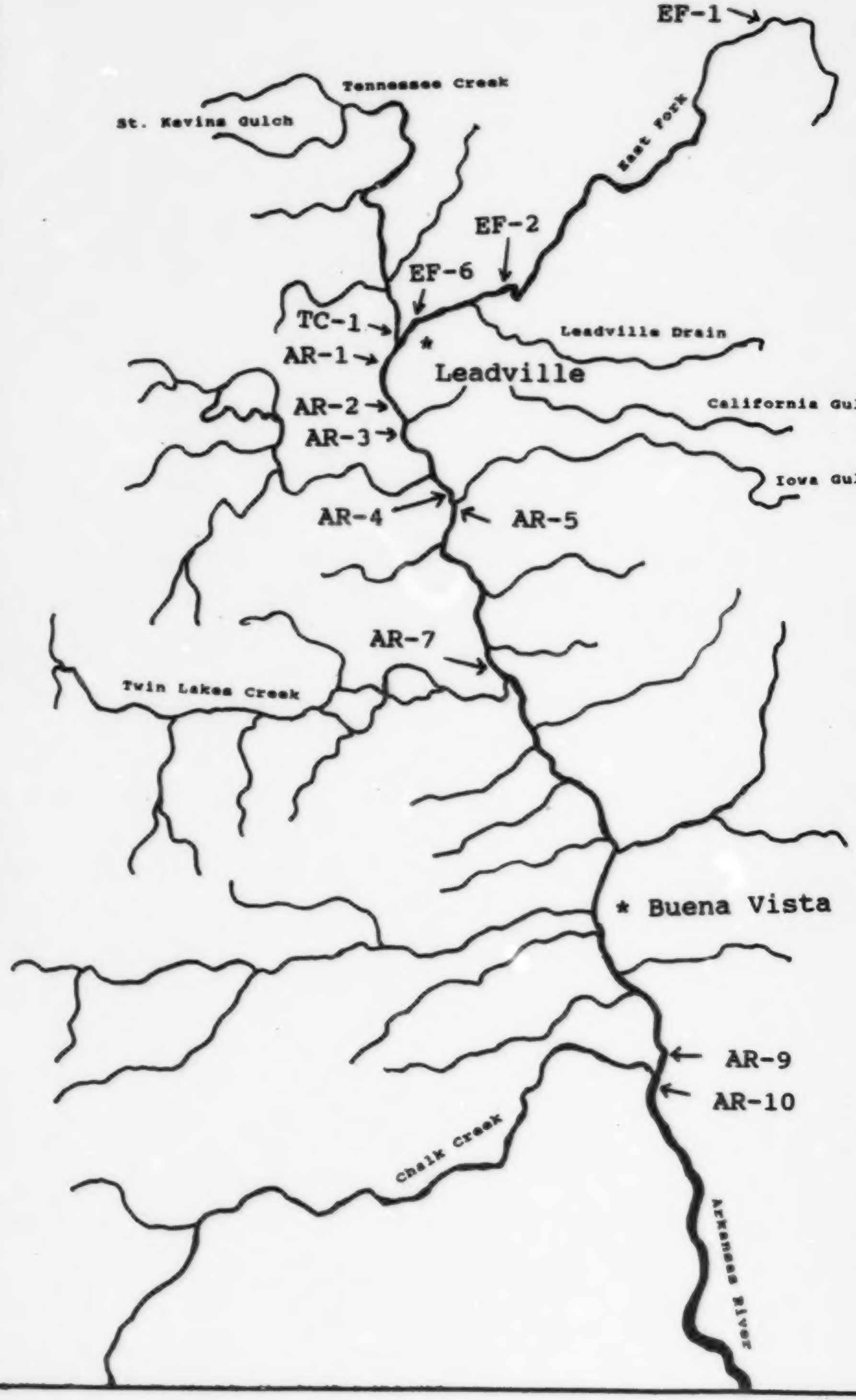


Figure 2. Arkansas River Toxicity Profile (September 22 - 30, 1987).

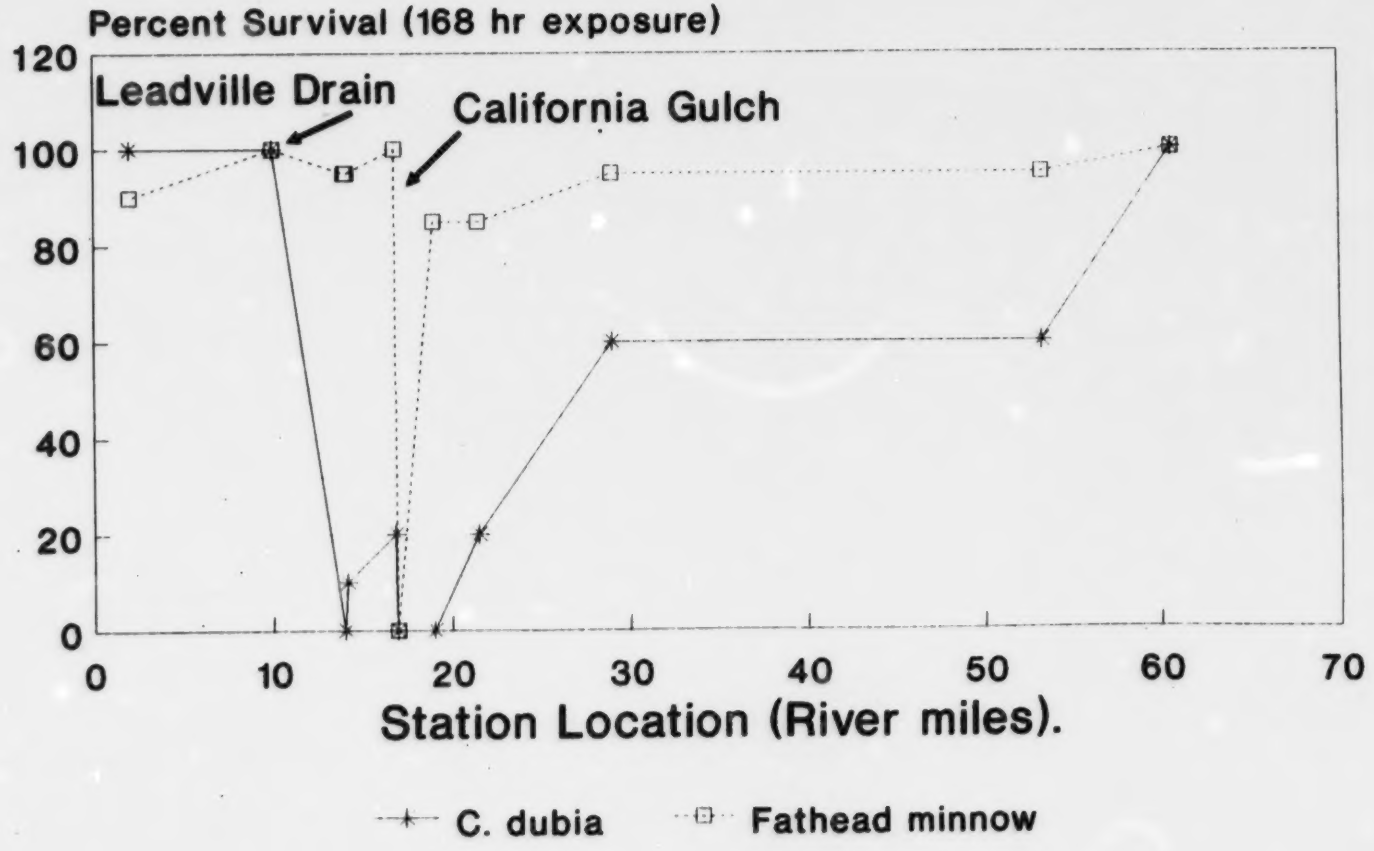
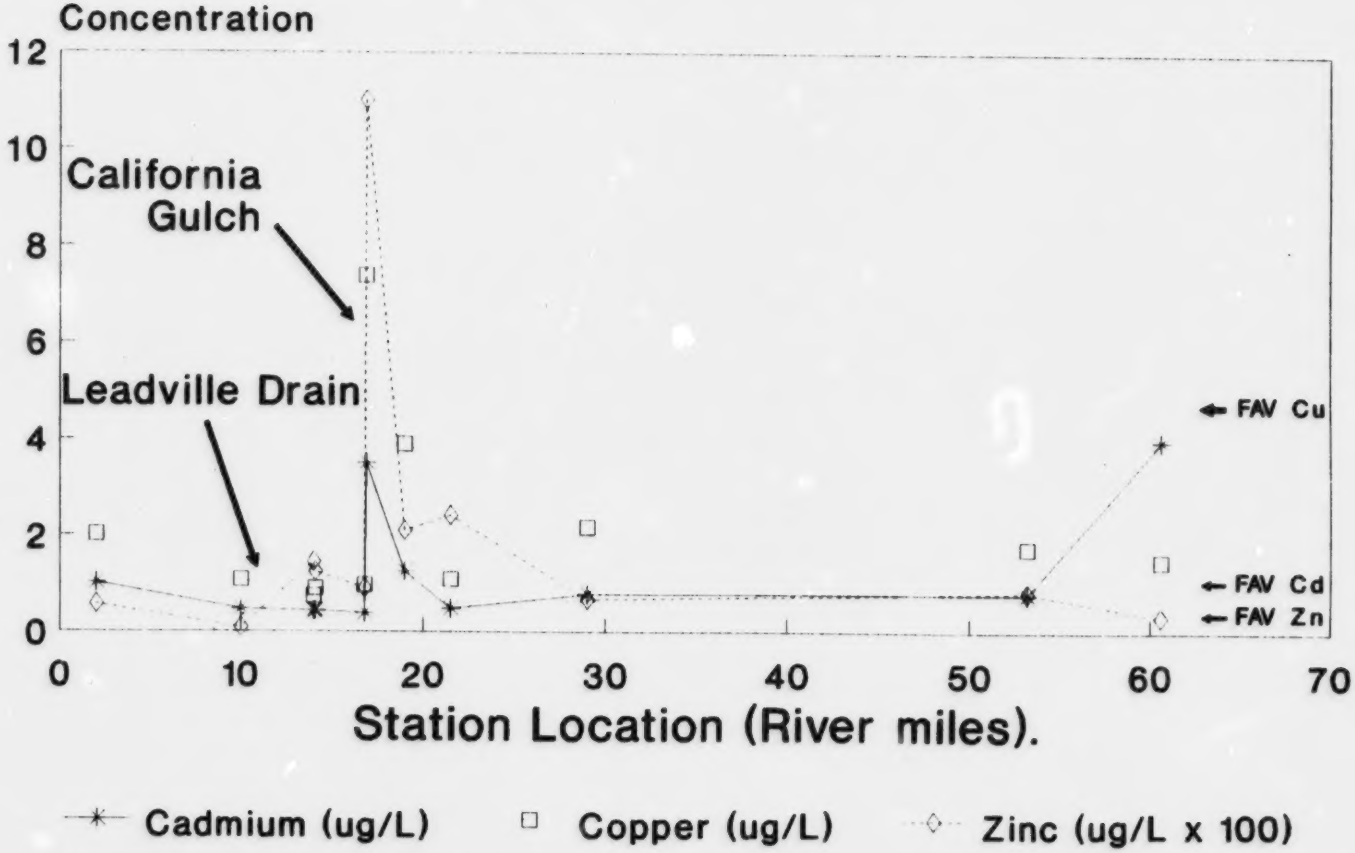
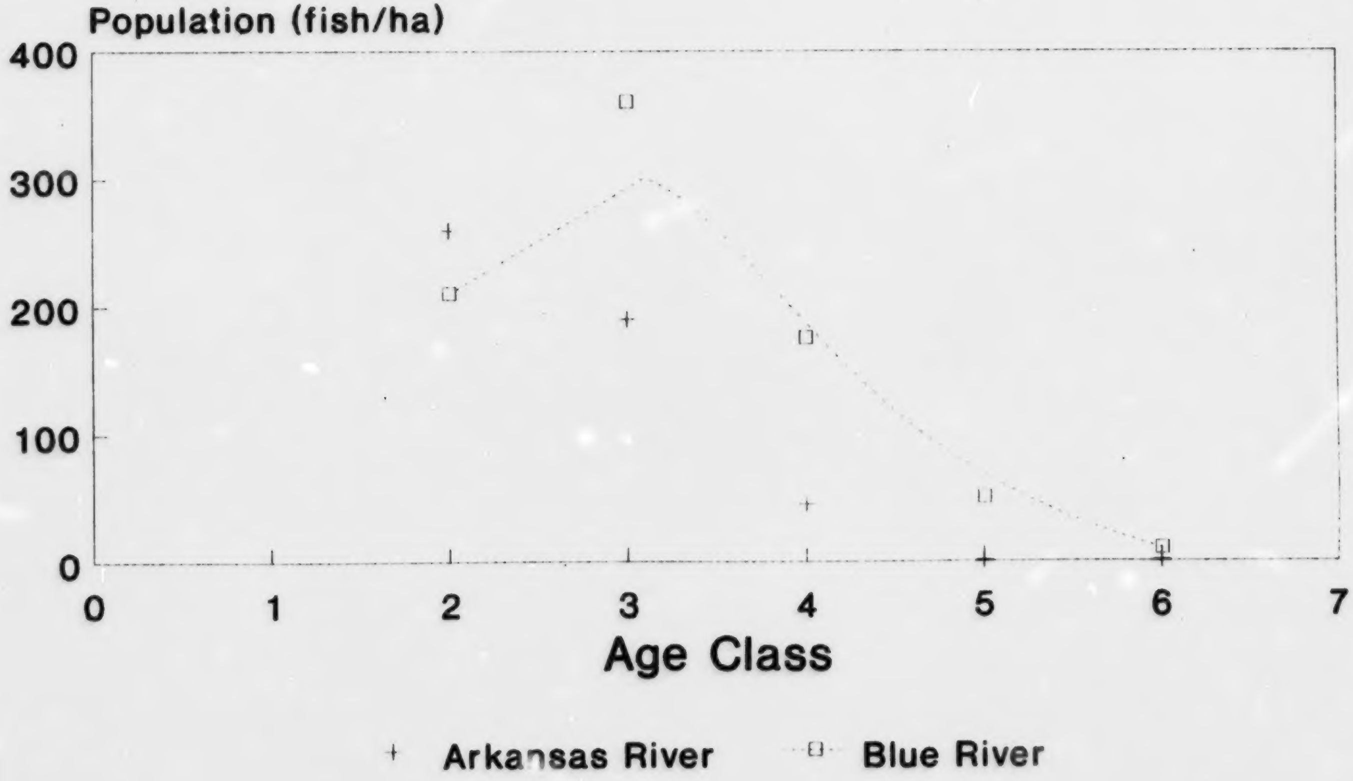


Figure 3. Arkansas River Water Quality Profile (September 22 - 30, 1987).



FAV = Final Acute Value (adjusted to hardness = 50 mg/L)

Figure 4. Brown Trout Population Structure in the Arkansas and Blue Rivers, Colorado (1981-1986).



Aquatic Mesocosms: Validating Water Quality Objectives

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The Water Quality Objective for cadmium in the Great Lakes, according to the Canada-US Water Quality Agreement, is 0.2 ug/L. This objective is based on the study by Biesinger and Christensen (1972) who reported a 16% reproductive impairment by Daphnia magna at 0.17 ug/L. A 16% reduction in reproduction was the lowest response that Biesinger and Christensen felt they could statistically distinguish from controls. The value of 0.17 ug/L was the lowest reported toxic response for any species at the time the objective for cadmium was written, and rounded off to 0.2 ug/L, formed the basis for the objective. Similarly, the objectives for most other metals, written by the Ecosystem Objectives Committee, or its predecessors, of the International Joint Commission, are also based on the lowest observed toxic response reported in the literature.

We repeated the study of Biesinger & Christensen (1972), but extended the exposure period to 8 weeks. We found that our animals were slightly less sensitive to cadmium (Borgmann et al. 1989a). This could have been caused because our study was done using Lake Ontario water, whereas Biesinger and Christensen's study was done using Lake Superior water, which has a lower alkalinity and hardness. Alternatively, we might have been using a genetically more resistant stock of animals. Based on our study, using the same procedures, we would set a water quality objective of about 1.0 ug/L for Lake Ontario. (Recent studies have indicated that Hyalella azteca is slightly more sensitive than Daphnia magna, demonstrating chronic mortality at 1.0 ug Cd/L in Lake Ontario water (Borgmann et al. 1989b). An objective of 1.0 would therefore be too high, but for the moment we want to consider only an objective based on Daphnia).

These experiments are typical of the single-species laboratory toxicity tests on which many Water Quality Objectives are based. An important consideration is whether such tests accurately estimate the sensitivity of the organisms in the field, where the animals are a part of a complex ecosystem with unknown multi-species interactions and biological feedback mechanisms. To answer this question we undertook a series of ecosystem level experiments in the laboratory.

Our ecosystems consisted of 4 large volume (3400 L) stainless steel tanks, 1 m in diameter and 4.5 m tall (Borgmann et al. 1988). Light, temperature, vertical mixing, nutrient addition rates, and zooplankton harvesting rates were maintained at constant levels for the duration of each treatment. The sole zooplankton species was Daphnia magna and the phytoplankton were dominated by Chlorella sp. Experiments lasting for up to 20 weeks demonstrated that these species could maintain stable populations for extended periods of time. Steady-state biomass levels for both phytoplankton and zooplankton were predictable from standard ecosystem models, as were the damped oscillations observed as the system approached equilibrium. Conditions of nutrient addition and zooplankton harvesting rates which would lead to unstable populations were also predictable (Borgmann et al. 1988). Results demonstrated that these systems were, in fact, stable ecosystems and not just transient populations.

Cadmium additions of 1, 5 and 15 ug/L were made to these ecosystems. After the first addition, cadmium levels were monitored and additional cadmium was added as required to maintain the cadmium at constant levels. Cadmium concentrations of 1 ug/L, the predicted safe level, had no effect (Borgmann et al. 1989a). Additions of 15 ug/L drastically reduced both phytoplankton and zooplankton populations after about 5 weeks. Levels of 5 ug Cd/L eventually resulted in an almost complete elimination of *Daphnia* and a dramatic increase in phytoplankton levels. Although cadmium is toxic to *Daphnia* at this concentration, these animals are still able to reproduce in benchtop toxicity tests, and a complete population crash was not anticipated. Furthermore, the crash did not occur until about 8 weeks after the cadmium addition. This dramatic decline in the *Daphnia* appears to result from an initial decline in *Daphnia* biomasses, which leads to reduced grazing pressure on phytoplankton and an increased phytoplankton biomass. Eventually, the phytoplankton biomass becomes high enough to inhibit *Daphnia* population growth, and a positive feedback loop results in the virtual elimination of *Daphnia*.

Three conclusions can be drawn from these experiments. 1. The benchtop bioassay correctly predicted the cadmium concentration which would result in deleterious ecosystem effects. The use of such toxicity tests in setting water quality objectives appears to be justified. 2. The nature of the response of the ecosystem, and its degree, cannot be predicted from benchtop bioassays. Positive feedback mechanisms can create dramatic population changes. 3. The time required to observe a response at the ecosystem level may be quite long (i.e. equivalent to several generations of the slowest growing animal). A lack of response at the ecosystem level after an objective has been exceeded for a short time does not necessarily demonstrate that the objective is lower than required to protect the ecosystem.

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Formulation of Water Quality Objectives for an Industrial River System: The Saint John River as a Case Study

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INTRODUCTION

The Saint John River basin is one of the largest watersheds in eastern North America and drains approximately 20 000 square miles of Maine, Quebec and New Brunswick. Stretches of the river form the international boundary between Canada and the United States and a number of its tributaries cross national or interprovincial boundaries. This paper briefly describes the progress to date, in the development of water quality objectives for the international sections of the Saint John River. These will be used by Canada and the United States as one of the management tools to assist in the promotion of activities to protect the quality of the river.

HISTORICAL INFORMATION

In 1971, the NATO Committee on the Challenges of Modern Society studying inland water pollution, selected the Saint John River basin as a case study for international cooperation. The Canada-U.S. Committee on Water Quality in the Saint John River was formed composed of the U.S. EPA, the State of Maine Department of Environmental Protection, Northern Maine Regional Planning Commission, Environment Canada and the New Brunswick Department of the Environment.

In 1974, a working group was set up to develop water quality objectives to the international portions of the river. The initial set of water quality objectives were not accepted by the Committee for the following reasons:

1) There was insufficient information on the existing water quality;
2) Some of the objectives were for parameters not considered controllable;
3) Objectives had been set for all uses, not just for the planned uses of the river, and
4) Too many parameters had been considered.

In 1979, a subcommittee, the International Technical Advisory Subcommittee was created to review and revise the original water quality objectives, taking into account the concerns expressed by the various participating agencies. The Subcommittee had members from Environment Canada, Fisheries and Oceans Canada, New Brunswick Department of the Environment, Quebec Department for the Environment, U.S. EPA, U.S. Geological Survey, Northern Maine Regional Planning Commission and State of Maine Department of Environmental Protection.

The Subcommittee developed a new set of water quality objectives and also formulated a monitoring and analysis program for the international sections of the river. The objectives were officially adopted by Canada and the U.S. in 1984.

Water quality objectives (for the uses "fishable, swimmable) were developed for seventeen parameters including dissolved oxygen, pH, temperature, mercury, zinc, copper and lead. The effects of physicochemical characteristics of the environment on the parameters of interest were taken into account, but synergism between parameters was not. Monitoring is done at a minimum of eight times a year and the data are compared to the "safe" limit. Each individual grab value is compared to the acute value in the rationale for the objective. Values must always be below the acute values, and if a parameter was frequently above the acute value it was dealt with as a special case. The values were evaluated to determine whether it was indicative of a real problem or was an anomaly. High levels could occur during spring run-off but were within limits for the rest of the year.

If concentrations of parameters (covered by water quality objectives) were continually below the analytical levels of detection, the development of new methods for analysis was recommended.

USE OF THE WATER QUALITY OBJECTIVES

The work on the Saint John River is done on a voluntary basis by the participating agencies, by sharing the work load as resources permit.

A number of special studies, in addition to regular monitoring, have been conducted in various reaches of the river and its tributaries in order to address specific problems. The Subcommittee analyses the results of all the studies and prepares an annual workplan, using the water quality objectives as one of the management tools for this analysis.

The water quality objectives and the monitoring data are used to:

- identify if present water quality can sustain specific water uses;
- identify the need for pollution control;
- assist in specifying effluent discharge requirements for the protection of specific water uses; and
- assess the effectiveness of pollution control measures.

SOME RESULTS OF THE INTER-AGENCY COOPERATION

The quality of the Saint John River is improving and further development of water quality objectives is anticipated. Industry views the water quality objectives as desired water quality that should to be aimed for.

FUTURE PLANS

A number of questions have to be answered in the future. There are plans to determine which are the toxic contaminants of concern and then to develop water quality objectives for them in the next two years. Management plans are to be refined and a study of non-point source pollution will be undertaken. Another study is planned which will determine whether the assimilative capacity of the river can be taken into account when developing water quality objectives.

Mutagenicity and Carcinogenicity:

Applications to Water Quality Objectives Development.

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PERSPECTIVE

Mutations, which are heritable changes in the base sequence of DNA, provide the variability in phenotype necessary for natural selection. While mutations are desirable in the context of populations, they may be viewed as deleterious to individuals because they can cause diseases. The spontaneous mutation rate of DNA may be in part due to natural mutagenic chemicals, derived from plants, animals and pyrrolysis (burning) reactions in particular, but synthetic mutagenic chemicals can increase this "natural" rate many times. Regardless of evolutionary considerations it appears to be in the public's, and the environment's, best interests to reduce exposure to synthetic mutagenic chemicals.

BACKGROUND

No data exists to suggest that mutations induced by man-made chemicals have been deleterious in the aquatic ecosystem. That is not to say that such mutations don't exist, rather that they have not been identified. Certainly chemicals with mutagenic activity have been identified in industrial discharges, and in particular effluents and chemicals discharged by the pulp-and-paper sector and steel industries can be mutagenic in laboratory tests (Table 1). However the observation of cancer (neoplasia) in populations of wild fish inhabiting polluted basins (Table 2) raises suspicions about carcinogenic discharges. Diseases such as neoplasia (and terata - misshapen organisms) which are observed in wild populations must be the driving force for the development of water quality objectives which consider mutagenicity and related diseases such as cancer.

The relationship between mutagenesis, carcinogenesis and teratogenesis is complex, however a generalization is that many carcinogenic and teratogenic chemicals are also mutagens. This has led to the development of short-term tests for mutagenicity which also then detect potential carcinogens and teratogens. Assays for carcinogenicity generally require exposure periods equivalent to or exceeding the age to maturity of the test organism, for example, several years for rats versus 6 months for the Japanese medaka, a small tropical fish. In contrast, short-term bacterial or cell culture tests for mutagenicity typically require only several days to weeks to perform. The difficulties in extrapolating from short-term in-vitro tests detecting mutagenicity to carcinogenicity, mutagenicity and teratogenicity in-vivo over an organism's life-span are many, but the time and cost advantages of in-vitro tests are considerable.

Because of the absence of more appropriate data extrapolation may be the only way in which the mutagenic and carcinogenic risks of a chemical in the aquatic ecosystem can be addressed. The polynuclear aromatic hydrocarbon Benzo-a-pyrene (BaP) is an example of an environmental mutagen and carcinogen for which aquatic data are available. Fish exposed to BaP in the laboratory (Table 3) and in the wild (Table 2) do develop cancer. Laboratory studies using IP (intraperitoneal) and egg injections and dietary exposure (Table 3) cannot be used to set water quality objectives. Benzo-a-pyrene is also teratogenic and genotoxic (causes non-heritable DNA damage) to aquatic species and causes mutations in bacteria if fish enzyme preparations are used for chemical metabolism (Table 3). These data are appropriate for criteria and can be used to recommend water quality guidelines to provide some protection against carcinogenesis. This type of approach, used by Environment Ontario, clearly utilizes available data for aquatic species whenever possible, rather than extrapolating from rodent feeding studies to aquatic species. Such water quality objectives may be interim measures until ecosystem objectives can be developed which consider factors such as contaminant flow between trophic levels, the relative impacts of sediment, water and dietary uptake rates and other complicating factors.

REGULATORY ISSUES

The evaluation of chemicals for mutagenicity and carcinogenicity in the aquatic environment is a new area, and before any attempts to analyze data are made a regulatory framework is needed. Do mutagenicity and carcinogenicity fall into a sub-category of chronic (sub-lethal) toxicity or are they sufficiently removed from the "mainstream" that they must be addressed separately? The fact that cancer and mutation induction are very specific endpoints rather than generalized effects (such as mortality, growth inhibition etc) indicates that they should be treated separately. Mutations are an endpoint which is sensitive and specific to a chemical's mode of action, covalent binding to DNA. It makes sense to regulate a chemical with a defined mode of action for their specific effect, and this may also include cancer, enzyme induction or inhibition, or teratogenesis. Mortality and growth inhibition may be due to general narcosis, but whenever possible chemicals with a specific mode of action should be regulated on that basis.

Establishing water quality objectives which consider mutagenicity and carcinogenicity may require new approaches to data acceptability, different from the normal objectives setting process. The principal issue hindering the development or setting of water quality objectives for mutagenic and carcinogenic chemicals is the absence of data for aquatic life forms. This is partly a result of the tremendous industry (U.S.) devoted to the detection of mammalian carcinogens. This has been prompted by concerns about food and industrial sources of carcinogens, generating very little interest in environmental contaminants, with the exception of those in drinking water. Much of the aquatic work in carcinogenesis has been aimed at replacing expensive and prolonged screening tests for mammalian carcinogenicity. These new tests utilize fish embryos injected with trace quantities of suspected carcinogens; such studies can be completed in 6 months. Indeed, most of the aquatic data on mutagenicity and carcinogenicity is for fish, with little consideration given to aquatic invertebrates and plants. The use of aquatic species (fish) is also being promoted in response to the objections of animal rights groups to mammalian studies, and studies are not generally designed to address water quality objectives. Agencies must be willing to stimulate the development of protocols for assessing mutagenicity and carcinogenicity in aquatic species.

Because most mutagenic chemicals are also carcinogens (and teratogens), and many carcinogens are also mutagens, objectives to protect against mutagenicity can address carcinogenicity. Environment Ontario's proposed objectives setting process will address data and effects which are mutagenicity related (cancer, terata) if the chemical under consideration is a proven mutagen, and the disease may result from mutations. This process will not address carcinogenic chemicals unless they are mutagenic. Establishment of objectives strictly for carcinogenicity are unlikely to be developed for aquatic organisms. Carcinogenicity data, from laboratory studies, for aquatic species are not available, and the cost and time of such tests likely will preclude the development of such data.

OBJECTIVES

Environment Ontario's proposed approach to mutagenic chemicals is outlined in Table 4, and is part of the revised objectives development process. This process will utilize data for mutagenicity and also data indicative of mutagenicity-related endpoints such as cancer, terata and chromosome damage. This broad approach will serve to identify data gaps, leaving no doubt about the type of information which must be generated for chemicals in the future, in order to address aquatic mutagenicity hazards.

This approach is based on the concept that mutations (changes in the DNA) of a cell in an aqueous medium are a function of the extra-cellular concentration of the reactive chemical and not of the origin of the DNA. In other words the DNA of an algal cell, a macrophyte cell, an invertebrate cell and a vertebrate (fish) liver or other cell exposed to the same extracellular level should develop the same mutations. What determines the differences in mutation frequency between organisms is the production of reactive metabolites, often by Cytochrome P-450 dependant monooxygenases. The use of mammalian liver homogenate (S9) in association with *Salmonella* bacteria (Ames test) to predict mammalian mutagenicity is in accordance with this philosophy. As an example, BaP is mutagenic to *Salmonella* with S9 from rat, mouse, hamster, human, rainbow trout and crayfish; the bacterial cell and DNA can substitute for the aquatic animal's own cells and DNA. Vertebrates are roughly similar in their levels and types of MFO's (mixed function oxidase), however invertebrates and plants differ greatly as MFOs are not present, and thus BaP will not be oxidized to the mutagenic form; though bioaccumulation can still occur. Bioconcentration factors are used to correct in-vitro results, for example with *Salmonella* and trout S9 which predict mutations for trout liver cells, to predict a corresponding ambient (water) level.

The specific details of this objective setting process are still being formulated, however toxicity, bioaccumulation and mutagenicity are being given equal priority. If sufficient data are not available to confirm the mutagenic risk of a chemical to a whole aquatic organism, a "tentative objective" may be set which can utilize in-vitro data predictive of whole body results. Variable application factors are used in setting objectives (1 to 0.1) depending on the adequacy of the data. Tentative objectives utilize the best available data and an uncertainty factor (0.01) which reflects the lack of data for whole aquatic animals. The best available data for many chemicals may be an Ames test with rat S9, an in-vivo genotoxicity test with mice, or a cell culture mutagenicity assay. This data can be used to set a tentative objective as seen with BaP. The use of fish S9 in an Ames test gives a LOEL (lowest observed effect level) similar to that found with rat S9 in an Ames test, or fish cells in culture (Table 3), suggesting mutagenicity in rodent systems is roughly representative of that in fish systems. Correction of in-vitro studies with fish cells or S9 with a measured BCF (2500) yields an estimated LOEL in-vivo of 0.1 ug/L. In-vivo studies (Levels 1 and 2) for genotoxicity vary widely in their sensitivity, but establish LOEL in the range of 0.1 ug/L, as does teratogenicity. The use of an uncertainty factor (0.01) establishes a tentative objective level for mutagenicity of 1 ng/L. This could be compared with the IJC objective of 10 ng/L and NYSDEC (New York State Dept. of Environmental Conservation) level of 1.2 ng/L.

The use of in-vitro data for setting mutagenicity tentative objectives may be necessary because almost no level 1 (see Table 4) data will be available for aquatic contaminants. BaP is an exception because it is considered a representative mutagenic and carcinogenic PAH and is used to calibrate and test many new methods and techniques using aquatic organisms. Using stringent data requirements (for example only in-vivo data) would result in "insufficient data" for virtually every chemical investigated. Where mammalian test data is all that is available, some jurisdictions may choose to establish a "safe" level based on chronic toxicity, placing an asterisk (*) to denote possible mutagenicity or carcinogenicity, but such an approach would not stimulate and promote the development of aquatic based data for such endpoints. This is the main thrust of Environment Ontario's approach to objectives for mutagenicity.

RESEARCH NEEDS

For many mutagenic (biologically reactive) chemicals, including BaP, bioconcentration factors predicted from physical characteristics in solvents (octanol/water partitioning) over-estimate the observed BCF's, likely because of chemical metabolism. Measured BCF's, including both parent compound and metabolites, will be more useful, but some estimate of the rate of generation of reactive metabolites by MFO's is what is really necessary. Clearly objectives do not at present address mixtures. Little is known about the mutagenicity or carcinogenicity of mixtures, but Black (1985) induced cancer in 8% of trout injected as eggs with 10 ug BaP while Metcalfe (1988) observed an 8% incidence with only 0.058 ug BaP when administered in a mixture (Table 3). The types and levels of MFO's in various species, and even the existence of MFO's in some species must be addressed in light of mutagenicity, and also the potential for biomagnification of compounds which are not metabolized.

CONCLUSION

Water quality objectives for mutagenicity and carcinogenicity may be driven by observations that epidemics of cancer and terata are found in some populations of wild animals, however the relationships between these endpoints are poorly understood. Regardless, appropriate (safe) levels for contaminants with the potential to cause these diseases must be established. We should rely instead on the spontaneous mutation rate of DNA to provide the raw material for natural selection.

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Table 1: Evidence for industrial and municipal sources of mutagenic and genotoxic contamination.

Source	Location	Sample (+/-)	Test	Reference
Pulp and Paper				
	Ocean coast	sed. ext. (+)	Ames *B. subtilis*	Kinae *et al.* 1981a
		fish ext. (+)	Ames *B. subtilis*	Kinae *et al.* 1981b
	Sweden	eff. ext. (+)	Ames	Kringstad 1981
	Can. (various)	effluent (+)	Ames CHO^{1}	Douglas 1980
		eff. ext. (+)	Ames CHO^{1}	
		constituents	Ames	
		constituents	Yeast	Nestman and Lee 1983
	Menominee R., Wisconsin	sed. ext. (+)	Ames UDS CHO^{2}	Fabacher 1988
	Fox R., WI	sed. ext. (+)	UDS CHO^{2}	Fabacher 1988
	India	effluent (+)	In-vivo MN	Das 1986
Chlorinated sewage				
	Sheep R. Alta.	Fish ext.(+)	Ames	Osborne 1982
		Sed. ext.(-)	Ames	
		Water (-)	Ames	Moore 1980
		Invert. ext(+)	Ames	
		Water X100(+)	Ames	
Wood Preserving				
	USA (unknown)	Sed. ext. (+)	Ames *B. subtilis*	Donnelly 1987a
		Sed. ext. (+)	*A. nidulans*	Donnelly 1987b
Steel Industry				
	Black R., Oh.*	Sed. ext. (+)	Ames UDS	West 1986
			UDS CHO^{2}	Fabacher 1988

Table 1 (cont'd):

Source	Location	Sample (+/-)	Test	Reference
	Cuyahoga R.*	Sed. ext. (+)	Ames UDS$_2$ CHO^2	Fabacher 1988
	Munuscong L.*	Sed. ext (+)	UDS$_2$ CHO^2	Fabacher 1988
	Hamilton Harb.*	Sed. ext. (+)	Ames	Metcalfe *et al.* 1988
Chemical Industry				
	Sweden	Eff. ext (+)	Ames	Sundvall 1983
Oil Refinery				
	Ontario	Eff. ext (+)	Ames	Metcalfe 1985
Miscellaneous				
	Rhine River	In-vivo SCE	Mudminnow	Alink 1980
	Rhine R.	In-vivo MA	Mudminnow	Prein 1978
	Rhine R.	In-vivo SCE	Mudminnow	Hooftman 1981a
		In-vivo MA	Mudminnow	
	California	In-vivo MN	Croaker Kelp Bass	Hose 1987
	Puget Sound*	In-vivo SCE	Sole	Stromberg 1981
		In-vitro AA	RTG-2	Kocan 1985
	Venice	In-vivo SCE	Mussel	Brunetti 1986
		In-vivo MN	Mussel	Brunetti 1988

* tumours found in fish from these locations

AMES. Mutations in *Salmonella*
CHO^1. DNA damage assay in Chinese Hamster Ovary cells
CHO^2. Chinese Hamster Ovary/HDPGT assay, for mutations in cell culture
UDS. Unscheduled DNA synthesis, indicates repair in cell culture
Yeast. Mutagenesis assay for reverse mutations with *Saccharomycescerevisiae*
B. subtilis. DNA repair assay with *Bacillus subtilis*
A. nidulans. Mutations in *Aspergillus nidulans*
SCE. Sister-chromatid exchanges
MA. Broken and abnormal chromosomes at metaphase (abnormalities)
AA. Broken and abnormal chromosomes at anaphase (abnormalities)

Table 2: Epidemiological studies finding cancer in wild fish populations which may be caused by chemical contamination of the water and sediments.

Location	Suspected agent	Species	Tumors	Reference
Hamilton Har.[1]	PAH's	Bullheads	Skin	Smith 1989 (in press)
			Liver	Smith (in preparation)
		Suckers	Skin	Smith 1989 (in press)
			Liver	Smith (in preparation)
			Skin	Cairns 1988
			Liver	Cairns 1988
Lake Ontario	?	Suckers	Skin	Smith 1989 (in press)
			Liver	Smith (in preparation)
Vancouver H.	?	Sole	Liver	Goyette 1986
Niagara R.	?	Bullheads	Liver	IJC 1987a
		Suckers	Liver	IJC 1987a
Munuscong B.	PAH's	Bullheads	Liver	IJC 1987b
		Walleye	Liver	IJC 1987b
Detroit R.	?	Bullheads	Liver	UGLCCS 1988[2]
		Walleye	Liver	UGLCCS 1988
		Suckers	Liver	Maccubbin 1988
		Bowfin	Liver	
Black R. Oh.	PAH's	Bullheads	Epid.	Baumann 1987
			Liver	
Cuyahoga R. Oh	PAH's	Bullheads	Skin	Baumann 1988
			Liver	
Boston H.	?	Flounder	Liver	Murchelano 1985
Chespeake Bay	?	Wh. Perch	Liver	May 1987
Puget Sound	PAH's,?	Sole	Liver	Malins 1985
River Elbe	?	Flounder	Liver	Peters 1987
		Ruffe	Liver	Kranz 1985
Hudson R.	?	Tomcod	Liver	Smith 1979

[1] Extracted sediments from Hamilton Harbour cause liver tumors in rainbow trout (Metcalfe *et al.* 1988. Can. J. Fish. Aquat. Sci. 45:2161-2167)

[2] Upper Great Lakes Connecting Channels Study p. 472

Table 3: Data pertinent to setting a water quality objective for Benzo-a-pyrene.

Endpoint	Species	modifiers	LOEL*	Level[1]	Reference
Mutagenicity					
	Salmonella	mix/rat S9	0.5	5	Metcalfe *et al.* 1988
	Salmonella	fish S9-UI	0.24	4	Milling and Maddock 1986
		AC	0.24	4	
		MC	0.24	4	
		rat S9-UI	-		
		AC	0.24	5	
		MC	0.3	5	
	Bluegill(BF2)		0.3	3	Kocan 1981
	Salmonella	fish S9-UI	0.2	4	Balk *et al.* 1982
		MC	0.2	4	
Genotoxicity					
	Trout-MN	larvae	0.21ug/L	1	Hose 1984
	Newt-MN	larvae	0.05	1	Jaylet *et al.* 1986
	Newt-MN	larvae	0.01	1	Grinfeld *et al.* 1986
	Newt-MN	larvae	0.01	1	Siboulet *et al.* 1984
	Bullhead-MN	IP	25 mg/kg	-	Metcalfe 1988
	Carp(2 sp),Tench-MN	IP	10 mg/kg	-	Al-Sabti 1986
	Mosquito Fish-SSB		0.1	1	Batel 1985
	Bluegill,Fathead-SSB		0.1 ug/L	1	Shugart 1988
	Trout(RTG2)-AA		0.05	3	Kocan 1985
			0.1	3	Kocan 1982
	Sole-SCE	IP	.5 mg/kg	-	Stromberg 1981
	Toadfish-UDS		0.25	3	Kelly 1985
	Mudminnow-MA		0.1 ug/L	1	Hooftman 1981a
	Killifish-MA		0.1 ug/L	1	Hooftman 1981b
	Carp(2),Tench-MA	IP	10 mg/kg	-	Al-Sabti 1985

Table 3 (cont'd):

Endpoint	Species	modifiers	LOEL*Level[1]	Reference
Cancer				
	Trout	dietary	1000 mg/kg -	Hendricks *et al.* 1985
		IP	10 mg/kgX12 -	
	Trout	IP-egg	136 mg/kg (10 ug/egg)	Black *et al.* 1985
	Trout	IP-egg	150 ug/kg - (14 ng/egg)	Metcalfe *et al.* 1988
Terata				
	Trout	larvae	0.21 ug/L 1	Hannah 1982
	Flatfish	larvae	0.1 ug/L 2	Hose 1982
	Trout	larvae	0.21 ug/L 1	Hose 1984

* Lowest Observable Effect Level (mg/L)

[1] see Table 4

UI= Uninduced S9, representing background MFO levels
AC= Aroclor 1254 induced S9
MC= 3-Methylcholanthrene induced S9
MN= Micronuclei
SSB= Single Stranded Breaks, detected by the alkaline unwinding assay
AA= Anaphase aberrations, chromosome damage seen at anaphase
SCE= Sister Chromatid Exchanges, abnormal exchange rates at metaphase
UDS= Unscheduled DNA Synthesis, abnormal repair rates
MA= Metaphase Abnormality, chromosome damage seen at metaphase

Table 4: Proposed structure for mutagenicity objectives setting; data types and application factors. All data must be generated by waterborne exposure, for mutagenic chemicals with demonstrated in-vivo effects and aquatic exposure data. This structure is intended to utilize as much as possible in-vivo aquatic animal data. In recognition that little information of this type will be available for many chemicals, in-vitro data can be utilized if a measured BCF is available so that the in-vitro level (mg/L) can be converted to body burdens (mg/kg) and hence to the relevant "safe" aqueous level (mg/L) (OMOE 1989).

Level	Type	Species	Application Factor	
			PWQO[1]	PWQG[2]
1	In-vivo	Freshwater aquatic species		
		vertebrate, invertebrate and plant	1.0	0.01
		any (1 or 2 species) of above	0.1	0.01
2	In-vivo	Saltwater aquatic species with freshwater relatives		
		vertebrate, invertebrate and plant	0.5	0.01
		any (1 or 2 species) of above	0.1	0.01
3	In-vitro	Intact cells of aquatic species, with or without endogenous activation from aquatic species		0.01
4	In-vitro	Intact cells or organisms (bacteria) with activation from aquatic species.		0.01
5	In-vitro	Intact cells or organisms (bacteria) with activation from any species, for example rodents.		0.01

The value resulting from the lowest LOEL times the appropriate application factor is compared with the toxicity and bioaccumulation preliminary PWQO's, and the lowest is selected as the final PWQO.

[1] Must have 3 separate in-vivo determinations including at least one genotoxicity or mutagenicity study to set an objective

[2] May include the use of BCF's to correct in-vitro data to estimated in-vivo exposure.

The Role of Water Quality Objectives in Environment Economy Integration

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INTRODUCTION

The purpose of this paper is to address how to ensure that the development of water quality objectives is in keeping with the changes that are rapidly taking place in environmental management. In particular, there is a need to consider implications of the commitment given by the federal and provincial jurisdictions in Canada to the concept of sustainable development. This paper briefly discusses the ideas of sustainable development, then addresses the role that has been identified for market forces to play in environmental protection, and finally outlines some implications of this for water managers.

Were environmental and economic planning truly integrated there would need to be less emphasis on developing standards for contaminant emissions and less reliance on environmental officers to enforce them - development would be more environmentally sustainable. One thrust of the new environmental agenda is focused on how to stimulate societal behaviour in order that enterprise profits from a healthy environment. A second is aimed at the replacement of the traditional paradigm of confrontation on this issue, for one of co-operation.

SUSTAINABLE DEVELOPMENT

The World Commission on Environment and Development, in its ground-breaking 1987 report, presented a perspective of our "common future" and set the stage for the broad acceptance in Canada of what we have come to call sustainable development. Since that time a number of activities have taken place in an attempt to institutionalize the ideas and spirit found in the Brundtland Commission report. These include: the creation and reporting of the National Task Force on Environment and Economy; the set up of provincial and a federal roundtable to discuss ways of implementing sustainable development; and a greater focus, by all levels of government in Canada, at both the political and agency level, on more forward-looking environmental policies and programs.

Sustainable development, also referred to as environmentally sustainable economic development, is essential if there is to be protection of the rights of future generations to the resources of the planet. It involves ensuring that the endowment we bequeath our children and our children's children includes no less (quantity or quality) of the planet's resources then we have received. The Brundtland Commission targeted world population growth and the economic activities of the industrialized world as the greatest threat to sustainable development. Clearly, the earth's population cannot continue growing without further crowding out other species on this planet, consuming vast amounts of natural resources and creating massive waste problems. There is a finite limit to the population that the world can sustain.

More relevant to the topic of water quality objectives is the criticism of the economic activities of the industrialized nations and the effect that these activities are having in limiting opportunities for future use and enjoyment. Over exploitation of forests, fisheries, petro-minerals, and wetlands has led to a reduction in the resource pool of the planet. But an equally serious stress affecting these resources has come from the polluting activities of society, by accident or intention, such as spills and effluent discharges. Thus, there is a need for a different approach and a different philosophy with respect to the dominance of relatively unhindered economic activities in our society.

The traditional approach to protecting aquatic ecosystems has been to set environmental objectives as the basis for effluent discharge concentration standards. Economic development and practices were unhindered so long as economic entities did not exceed these concentration standards. However, there is no incentive to pollute less than these standards allow or to even consider further research of non-polluting technology if the standards are being met. On the one hand, the standards have provided a level playing field for economic interests, but on the other hand, serve as a ceiling or upper limit to pollution control. Since the standards are specified by contaminant concentration, total mass loadings to a water body may end up exceeding the assimilative capacity of that ecosystem.

This process of setting standards has led to a reliance on regulation for enforcement. The implementation of sustainable development, as viewed by Brundtland and the National Task Force, did not downplay that approach, but did address new approaches to environment and economy integration emphasizing economic carrots as well as regulatory sticks.

ENVIRONMENT ECONOMY INTEGRATION

Better integration of environmental concerns into the economic framework of societal planning, as an approach to the achievement of sustainable development, involves necessary rewards and penalties for appropriate environmental behavior. Many argue that this could also come about by a change in the way that society philosophically regards natural resources - no longer free goods or unlimited in supply. Indeed, such an attitude change should come about through efforts, currently underway, to educate newer generations on the need for conservation to balance the exploitation of resources. Nevertheless, there is also a need for market place stimuli involving penalties and rewards to motivate immediate change for unsustainable activities.

The National Task Force on Environment and Economy, in its September 1987 report, recommended that "Government, industry, academic and other non-government organizations should develop new tools and improve existing tools which achieve more efficient and effective environment-economy integration". They went on to highlight examples of "economic mechanisms such as contaminant charge schemes, tradeable emission/discharge rights, financial assurance and performance deposits, investment tax credits, credits for exceeding environmental standards, and reduced interest bonds", and to call for "improved techniques for the valuation of environmental stresses and the benefits of environmental protection" and "economic incentives which promote effective environmental protection by business".

That which has commanded the interest of the business sector in sustainable development has been the prospect of adopting more environmentally friendly practices through the provision of new economic incentives. The goals of industry have been such that short term profits, and not the environment, are paramount in planning decisions. Just as penalties are necessary to enforce pollution regulations, and the greater the penalties the greater the attention paid to compliance, economic incentives are necessary for industry to change production processes and production. Environment economy integration, at its most elemental level, means that industry must see the impact of corporate environmental policies on its bottom line. The role of government is to provide for and ensure that environmental behavior incentives are commensurate with environmental objectives.

The two approaches to achieving water quality improvements are to set and enforce water use and effluent emission standards and/or to provide incentives for society to use resources in a less polluting fashion. The latter approach places the role of economics centre stage in the process of environmental management and requires market interventions to ensure that the many non-traded natural resources reflect their real value to the global ecosystem and that the cost of polluting activities are internalized into the spreadsheet of the polluters. For example, water will be treated more like other resources if it is appropriately priced. This approach must be accompanied by a willingness to cast off the presumptions, so often raised about what society will and will not accept with respect to the lifestyles of individuals, and the corporate policies of individual firms. History has shown, in times like the so-called energy crisis, that society can and does adapt for the appropriate causes, given the right stimuli.

1. Water Demand Management

Higher prices for water will lead to reduced demand, less water used, and other things being equal, less pollution. The environmental implications of water conservation have not yet been fully explored, but initial studies have favoured the intuitive assumption that the less used, the less polluted. A study of "In-Home Conservation and Wastewater Management" by W.J. Hopp and W.P. Darby (1981) indicated that a ten percent reduction in water use, with required modifications to the treatment process, would lead to an equivalent 10% reduction in total biochemical oxygen demand and total suspended solids loading. This supported work by Bohac and Sierka (1978), who found that there was no indication that increasing wastewater strength, while proportionately decreasing wastewater flow, would impair an activated sludge plant's ability to meet a mass loading discharge requirement.

Higher prices can affect behaviour not only of household residents and commercial establishments but also of industrial water users. An industry that withdraws water directly from a water body or aquifer and pays no royalty has no incentive to use water wisely and a powerful economic incentive to use dilution as a method of meeting wastewater discharge standards. In general, the belief, cultured in our society, that water is an abundant free good has hindered the development of water efficient fixtures and practices in North America when compared to European and other societies and has contributed, no doubt, to the high per capita level of pollution in North America.

2. Water Pollution Ceilings

Approaches to wastewater management which identify and specify so-called "best available technology" (BAT) or "best available technology economically achievable" (BATEA) for use in wastewater treatment can lead to unsustainable levels of water degradation at great cost to the polluting industries for a number of reasons. New and better technology may not get looked at for years as society gets saddled with "the technical fix of the time". There is no incentive for industry, with a fully capitalized technology, to change when new technology is developed. The responsibility for waste gets shifted to governments, the BATs become institutionalized and it becomes more difficult, institutionally, to introduce new ideas and approaches. Since the control orders come from government, the costs are seen as being imposed by governments, and they end up paying. Research and development on production processes and waste reduction technology become a responsibility shared with governments and may get ignored for any number of years once a technology gets universal acceptance and installation. Old facilities, that were part of the justification for BATEA compromises on waste treatment, get replaced by new facilities, which are now constrained by the dated BATs - or they do not get replaced at all.

An alternative, and one that economists prefer is for environmental scientists to determine the degree to which ecosystems can withstand contamination/depletion, and to let the "market" allocate quotas for allowable pollution/exploitation to all users within the ecosystem. A ceiling; eg. 10 kilograms per day of phosphorous, for a water body would be allocated amongst users on the basis of how much each would pay for the right to pollute. It is important to recognize that this same right to pollute is currently given through control orders and standards set by concentration of effluent etc. Those with a small or no allocation would be forced to buy rights from others, to redefine processes in order to comply with their allocation, or to cease operations and relocate their production process to where an allocation is available. Water quality objectives, in this case, can be focused solely on ecosystem assimilative capacity and not in consideration of how to achieve it or the consequences of non-compliance - the market place efficiently takes care of that.

SUSTAINABLE WATER QUALITY OBJECTIVES

In an industrial market economy an efficient way to integrate environmental and economic interests is to ensure that the economic incentives for responsible environmental behaviour are in place at the market level. Environmental scientists should not second-guess the responses of industry and society to the need for a clean and healthy environment. If the right environmental objectives are in place and the opportunity exists for society to respond in a rational fashion, sustainable development with respect to water resources is attainable. There are, however, some immediate actions that should be considered in this forum on water quality objectives.

1. Persistent Contaminants:

Since persistent contaminants are long lived and therefore affect future generations, their presence in the aquatic environment is in conflict with sustainable development. It is, thus, a mis-allocation of scarce scientific resources to devote them to the development of water quality objectives or standards for these substances. Their presence in water bodies at any level is undesirable since they cannot be assimilated as part of the natural ecosystem. Future generations will inherit poisoned waterways regardless of the amounts of these contaminants that get released. Zero discharge and virtual elimination are the standards and water quality objectives for persistent contaminants.

The economic significance of "no discharge at detectable levels" for persistent contaminants could be dramatic since the great majority of halogenated compounds, heavy metals, and other problem substances fall into that category, and currently have few readily available alternatives in either products or processes. Since a similar rationale exists for zero discharge of these substances on land, into the air and through deep well injection this could lead to minimal or no production. The development of new products and processes, the investment in research and development, and the changes required in attitudes associated with consumption of these products will all result in economic impacts, many of which will be positive. There is currently an effort being made to examine the extent of chlorinated compounds in our society and the feasibility and economic impacts of their withdrawal from the market place.

2. Assimilative Substances

Essential trace metals, nutrients and other substances, which naturally occur and can be accommodated in natural water courses, are another class of substances, and ones for which the traditional approach of developing guidelines and objectives and setting discharge standards is not inconsistent with the theme of Brundtland. Long term sustainability places the responsibility clearly on water quality objectives to acknowledge the potential for ecosystem deterioration with increases in population growth, possible climate change effects, increased water consumption, greater shoreline development, fish re-stocking programs, and time. Secondary and indirect as well as acute effects should be considered as the basis for developing these objectives.

The environment-economy integration approach to water quality will best take place if environmental aspects are developed with a focus solely on the environment, and the economic activities are altered/adjusted and developed to reflect the need of the environment as well as the consumers and the corporate bottom line. Under no circumstances should water quality objectives be set "in a compromise" to permit the continued operations of industrial enterprises. A recent paper by W. Sinclair, in examining the actions of the pulp and paper industry with respect to technological change shows clearly that, at times when industry could have financially supported implementing less polluting technology, it neglected to do so.

SUMMARY

There are four messages to be found in sustainable development that relate to environment economy integration and water quality objectives:

1. Water quality objectives (aquatic ecosystem health) must be set on the basis of long term ecosystem needs - there should not be any attempt to second-guess the ability of society to meet the objectives in this setting process.

2. Objectives and standards for persistent contaminants are zero.

3. Objectives should be based on a total, ceiling or mass loading basis for non-persistent contaminants at specified sites and entire water bodies.

4. New approaches to water use management, which focus on economic incentives, disincentives, and other market interventions by governments, must be initiated to complement the traditional protection/enforcement approach.

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Managing water resources: Integrating Wastewater Treatment Technology with Water Quality Objectives

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INTRODUCTION

In 1981, the Ontario Ministry of the Environment began a study of the water quality in the Don and Humber Rivers and Mimico Creek to provide baseline data to guide future studies. The objectives of the study were to define the water quality conditions within the study area; to analyse the cause and effect relationships for problem constituents and areas; and to develop cost-effective measures for controlling pollutant loadings to the study area's receiving waters based on watershed needs and users. The studies were managed by a steering committee with representatives from the provincial, regional and municipal governments and conservation authorities. Public comment was sought on the proposed programs.

The response of water quality to remediation of a variety of sources (spills, erosion control, sewer use bylaw, agricultural control) was evaluated qualitatively. The potential response of aquatic toxicity and the fishery to remedial and mitigating measures were evaluated with quantitative models, but interpreted qualitatively. Special emphasis was placed upon the impact of spills remediation and upon integrating remediation into an ecosystem approach involving water quality, public health, public safety, aquatic toxicity, the instream fishery riparian vegetation and terrestrial habitats for water fowl and wildlife.

A strategy for improving water quality was developed, and several major works were undertaken, particularly in the area of sewage treatment. As the Municipal Industrial Strategy for Abatement of Pollution (MISA) does not involve stormwater, a program for implementing and auditing the achievement of water quality objectives for stormwater was developed. Other programs related to the development and use of water quality objectives, for example for aesthetics and fisheries.

Full details of the basin studies and the use of water quality objectives in an integrated program can be found in "Strategy for Improvement of Don River Water Quality" September 1989, a report prepared for the Steering Committee Toronto Area Watershed Management Study and obtained from the Committee c/o Environment Ontario, Water Resources Branch, 135 St. Clair Avenue W., suite 100, Toronto, Ontario, M4V 1P5.

Development, Implementation, and Use of Site-specific Water Quality Objectives: A Conceptual Model.

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ABSTRACT

The Canadian Council of Resource and Environment Ministers (CCREM, 1987) developed and published the Canadian Water Quality Guidelines to provide information to aquatic resource managers, industry, and the public. These guidelines recommend characteristics of aquatic ecosystems that are required to support and maintain designated water uses. While these national guidelines provide an excellent source of information on use requirements, there are many situations that require additional data to make rational decisions on how aquatic resources should be used. In these cases, site-specific water quality guidelines and/or objectives are needed. This paper outlines a conceptual model that may be used to facilitate the development of these management tools.

INTRODUCTION

Rational resource management is, perhaps, one of the most challenging undertakings associated with the field of environmental science. It requires the integration of the diffuse interests of resource user groups with detailed scientific information on the current strength, sensitivity and value of common property resources. Water quality management is particularly demanding because water users have interests that are not only different, but in many cases diametrically opposed.

Beneficial uses of water include those involving domestic water supplies, recreation and aesthetics, fish and aquatic life, agriculture, and industrial supplies (process water). However, our water resources are also used as sinks for domestic, agricultural, and industrial waste products. Inputs of sewage and other domestic wastes, pesticides, herbicides, heavy metals, and other toxic compounds can contaminate water resources, and compromise downstream water uses. The charge of water resource managers is to ensure that water resources are used wisely, to the greatest common benefit of all user groups.

Water quality managers require information on the water quality requirements of various water uses in the system under consideration to make rational decisions regarding the allocation of resources. According to the definitions of the Canadian Council of Resource and Environment Ministers (1987), some of the tools available to water quality managers include:

1. Water Quality Criteria - These are the scientific data that are evaluated to derive the recommended limits for water uses. For example, the 96 hour LC_{50} of un-ionized ammonia to 52 gram rainbow trout (pH = 7.88, T = 10.0° C, DO = 9.4 mg/L) has been reported to be 0.484 mg/L (Thurston and Russo, 1983).

2. Water Quality Guidelines - These are numerical concentrations or narrative statements recommended to support and maintain designated water uses. For example, no harmful effects on fish and aquatic life will result if un-ionized ammonia levels (pH = 8.0, T = 10° C, D.O. = 8.0 mg/L) remain below 0.025 mg/L (Canadian Council of Resource and Environment Ministers, 1987).

3. Water Quality Objectives - These are numerical concentrations or narrative statements which have been established to support and protect the designated uses of water at a specified site. For example, The average concentration of un-ionized ammonia in Howell Creek should not exceed 0.008 mg/L. This objective is predicted to protect sensitive fish and aquatic life species, and in so doing should ensure that other beneficial uses of water will not be impaired (MacDonald et al. 1987). Establishment of a water quality objective requires agreement between all of the agencies responsible for the management of water quality in the basin under consideration.

4. Water Quality Standards - These are water quality objectives that are recognized in enforceable environmental control laws of a level of government. For example, the average concentration of un-ionized ammonia in Howell Creek shall not exceed 0.008 mg/L.

These definitions are imperfect, and create some level of confusion for resource managers. The definitions are imperfect because they are not concise and require interpretation by the reader. The confusion stems primarily from the fact that in many jurisdictions guidelines are commonly referred to as criteria. Also, the meaning and utility of water quality objectives are not clear to many environmental managers. However, the advantage of using these definitions is that resource and environment agencies across Canada have agreed to use them, and that, in itself, contributes to their validity.

The purpose of this paper is to present a conceptual model that outlines a process that may be used to develop water quality objectives. In addition, the role of water quality objectives in the water quality management process will be discussed in the context of environmental risk assessment.

BACKGROUND

Interactions between water and human activities are so pervasive that water is considered a vital element in socio-economic development. These interactions involve a wide range of essential biological, social, and economic functions, as well as negative functions such as flooding and disease transmission (Cox 1987). Indeed, so profound is the link between water and development that there is a continuing effort to enhance the positive attributes of water, while trying to eliminate those negative aspects. Water management is an imprecise art that attempts to coordinate the interactions between water and human activities to the greatest common good of society as a whole.

The goals of water management are determined, to a greater or lesser degree, by the users of the water resource. In Canada, where water is largely in good supply, water courses have been viewed simply as conduits to carry away effluents, to impound for hydro-electric power, or to remove excess water from agricultural lands. Immediate human demands have usually been placed at the forefront of management policy, and have traditionally included water for domestic, industrial, and agricultural consumption. Other beneficial uses of water, such as those associated with fish and aquatic life, have generally been given a secondary importance.

In recent years, additional demands have been placed on water resources. Increasingly, water resources are being used as sinks for highly organic toxic chemicals, such as chlorophenols and dioxins from the lumber and pulp and paper industries. At the same time, inputs of pesticides and herbicides from agricultural sources have not diminished. The mining industry continues to release significant quantities of toxic metals and cyanide to stream systems, while acid mine drainage is rapidly becoming one of the most serious issues facing environmental managers. In addition, non-point sources of pollution arising from agricultural and forest management activities are superimposed on those sources that are more quantifiable.

On a more positive note, public awareness of the impacts of developmental activities is improving. There is an increasing level of concern over the purity of drinking water supplies. The maintenance and enhancement of high quality fisheries has been identified as a high priority, long-term goal, and habitat conservation and improvement has been shown to be a critical factor in achieving this goal. Additionally, recreation and aesthetics is an emerging water use that will require more consideration in the future. How then, in the midst of all these competing demands, can water quality managers make rational, defensible decisions about the manner in which water resources will be used?

In the past, and to a decreasing extent today, water managers have relied on surpluses of water to meet the demands of various user groups. Under these conditions, decisions regarding water resources have been made on an ad hoc basis. Developers have applied for, and received, permits to discharge effluents into receiving water systems. Levels of waste treatment have been highly variable, and have generally been related to importance of downstream water uses. In some cases, state of-the art technology has been required to meet management goals. In other cases, particularly in remote locations, waste treatment has been non-existent (for example, in placer operations). Environmental conservation and protection has not been a serious concern. Only in circumstances where public drinking water supplies were at risk have conflicts over water use been in evidence. And, for the most part, identified risks have been associated with the transmission of pathogenic organisms.

This management strategy is no longer adequate. Today, water management decisions must be defensible and objective. Achievement of these goals necessitates characterization of the requirements of the various water uses in terms of water quality. Integration of this information with data on ambient environmental, social, and economic conditions provides a basis for rational decision making. Effective water management also requires political support for decisions that are made in a rational, defensible manner.

CONCEPTUAL MODEL

Water quality objectives, formulated on a site-specific basis, provide a framework for making scientifically and economically defensible decisions on how water resources ought to be used. This paper provides an overview of a water quality objectives development process. In addition, it discusses how water quality objectives can be used within the existing regulatory framework to affect sound water quality management. The model presented in this paper has been organized into seven distinct units or modules that illustrate the various components of the overall process. These modules must be integrated (Figure 1) to provide the necessary information. The fundamental components of the model include the following:

1. Regional Basin Assessment
2. Data Collection and Interpretation
3. Theoretical Toxicological Assessment
4. Applied Toxicological Studies
5. Water Quality Objectives Development
6. Compliance Monitoring
7. Environmental Impact Assessment

Brief descriptions of each module are presented in the appropriate sections of this paper. The methodology presented applies specifically to the development and implementation of water quality objectives pertaining to the conservation and protection of fish and aquatic life. With modification, however, the model provides a useful general approach to the development of water quality objectives designed to protect and maintain other water uses.

1. Regional Basin Assessment

Development of site-specific water quality objectives for a body of water requires, by definition, detailed information on the river basin under study. The Regional Basin Assessment (Figure 2) is the process whereby the available information on the water body is collected, collated and analyzed to identify existing and potential uses of water, to assess ambient water quality, to identify sources of pollution, and to identify the potential pollutants expected from future developments. In addition, the available data is critically reviewed to determine its applicability and completeness. Subsequent screening of the available information using water quality guidelines, such as those prepared by Provincial agencies or the Canadian Council of Resource and Environment Ministers (1987), facilitates the identification of priority water quality variables with respect to protecting and conserving designated water uses. The thoroughness and accuracy of the regional basin assessment will, to a large extent, determine the applicability and appropriateness of the water quality objectives developed later.

2. Data Collection and Interpretation

A thorough Regional Basin Assessment not only provides the information required to identify priority water quality variables, but also clarifies the need for additional physical, chemical, and biological data. It is, therefore, more likely that new survey and monitoring programs will be well focussed and provide the types of information required later in the objectives development process. Requirements for physical data may include such variables as water temperature, stream-bed substrate quality, or information on riparian habitats (Figure 3). Baseline data for priority water quality variables should include estimates of temporal and spatial variability. Biological data collection will usually focus on determination of the timing, distribution, and abundance of important ecosystem components. In addition, measures of species composition and diversity are important to assess later changes that might occur due to anthropogenic activities.

3. Theoretical Toxicological Assessment

The Theoretical Toxicological Assessment (Figure 4) is one of the most important components of the water quality objectives development model. In many situations, this assessment will provide the information used to establish the water quality requirements of the most sensitive water use. It, therefore, forms the scientific basis for site-specific water quality guidelines and subsequently, the site-specific water quality objectives. The quality of the work done during this assessment will, in a large measure, determine the value of the objectives formulated.

There are a myriad of approaches that may be used to expedite a theoretical toxicological assessment, but there is a common thread that runs through them all. Each of these techniques relies on the premise that aquatic toxicology information reported in the literature can be applied to the system under study to facilitate predictions about the significance of various toxicants in freshwater ecosystems. Prediction of safe or no effect levels of the toxicant is the most common prediction made, but many other predictions are also possible.

The simplest approach to the toxicological assessment is to rely on general guidelines that have been developed for the protection of a beneficial water use. These guidelines (also called criteria in many jurisdictions) range from rather general narrative statements to rather complex numerical derivations that require some site-specific information. Some of the more comprehensive guidelines produced relating to freshwater fish and aquatic life include; (i) Canadian Water Quality Guidelines (Canadian Council of Resource and Environment Ministers 1987), (ii) Water Quality Criteria for Freshwater Fish (Alabaster and Lloyd 1982), (iii) Water Quality Criteria for European Freshwater Fish (European Inland Fisheries Advisory Commission 1987), (iv) Water Quality Criteria 1972 (National Academy of Sciences, and National Academy of Engineering 1973), and (v) Ambient Water Quality Criteria (Environmental Protection Agency 1980 -1985). Unfortunately, all of these general guidelines, by definition, lack the specificity required to make defensible predictions for the unique site under consideration. Guidelines developed by provincial and state agencies for specific variables are applicable on a regional basis, and are therefore more likely to be relevant to particular sites. Notwithstanding their lack of specificity, all of the general guidelines are valuable tools, particularly for screening water quality and related data from the system under study to help identify priority water quality variables.

More complex approaches involve detailed reviews of the current scientific literature, with an aim to develop generalized relationships between a toxicant and other environmental factors. These relationships are then applied to the basin under study by using the site-specific data generated during the Regional Basin Assessment and the Data Collection and Interpretation modules.

The quality and quantity of site-specific information used in the Theoretical Toxicological Assessment will determine the confidence in the final acute and chronic values generated.

4. Applied Toxicological Studies

Theoretical Toxicological Assessments can provide a great deal of the information required to develop scientifically defensible water quality objectives. However, for many water quality variables there may not be enough aquatic toxicology information to perform an exhaustive assessment. In other cases, the applicability of laboratory studies in assessing ecosystem responses might be questioned. Cairns (1983, 1986a, 1986b) presents a number of convincing arguments for the multi-species or validation approach to environmental toxicology. These arguments include to following:

A. Even the most meticulous single species laboratory bioassay test cannot accurately predict how several such species might interact competitively or as predator-prey in the natural environment.

B. Because the accuracy of prediction of laboratory toxicity tests is generally questionable, large safety or application factors are used to adequately protect the resource. This approach can lead to large over or underestimates of toxic effects on the aquatic environment.

C. From a holistic point of view, even the 'most sensitive species' concept is flawed since new properties are added to systems as components interact.

D. Validation of laboratory bioassay tests is the ability to predict the relationship between the response of the artificial laboratory system and the natural system. Therefore, the validation process will be simpler and more direct if the laboratory tests are carried out on the same response that will be used for validation in the natural system.

It is apparent, then, that the accuracy of the prediction of responses to exposures of aquatic biota to environmental pollutants is greatly increased by collecting detailed toxicological information on the system under study. These studies may be as simple as 7-day static-renewal tests using site water to assess Ceriodaphnia acute and short-term chronic survival, growth, and reproduction. More complex bioassays might be run on resident fish species over their life history to assess long-term chronic effects on sensitive end-points like the incidence of brain lesions. Even more intricate tests involve the construction of mesocosms or whole ecosystem manipulation. The costs associated with these studies range from reasonable to outrageous; however, these expenditures are justified when a great deal of confidence in predictions of responses is required. When aquatic resources are deemed to have exceptional values (such as those associated with the Fraser River Estuary) or when the cost of secondary or tertiary waste treatment is prohibitive then the costs associated with applied toxicological studies (Figure 5) are easily justified.

5. **Water Quality Objectives Development**

The preceding modules have outlined the information required to develop site-specific water quality guidelines and assess how water uses are affected by existing or future water quality in the basin under study. These guidelines, to a greater or lesser degree, incorporate site-specific information to increase their applicability. Use of safety factors and application factors further increases the likelihood of use protection. However, political and economic interests may dictate that some compromise between use protection and socio-economic goals be achieved. These interests should surface and be addressed during the water quality objectives negotiating process (Figure 6).

Once the water quality objectives have been negotiated and implemented, water managers must decide how these objectives will be used to manage the aquatic resources under their jurisdiction.

6. **Monitoring for Compliance with Water Quality Objectives**

For water quality objectives to be useful to water managers they must be complied with and they must protect and conserve water uses. Assessment of the level of compliance with adopted water quality objectives necessarily requires implementation of a monitoring program that is designed to detect exceedances of the objectives (Figure 7). The frequency and timing of the sampling will be dependent on temporal and spatial variability of the environmental quality variable under consideration, the critical periods of water uses (eg. spawning periods), and on the nature of the developmental activities in the basin.

The monitoring program must also contain aspects that enable the resource manager to assess the health of the aquatic ecosystem and the maintenance of water uses. This may be as complex as detailed biological sampling to assess the status of population and ecosystem variables, or as simple as comparing water quality monitoring data to published guidelines for domestic water supplies. In situations where the objectives are not being complied with, jurisdictional action is warranted. In other instances where water uses are not being protected even though the objectives are being complied with, the objectives must be modified.

7. Environmental Impact Assessment

The Environmental Impact Assessment process (Figure 8) is designed to identify and resolve potential adverse environmental effects of a proposed development project. The process is based on the assumption that the potential effects of a development can be predicted using information readily available to the review agencies. The water quality objectives development process is supportive of the Environmental Impact Assessment because it generates detailed cause/effect information relevant to the system under study. Integration of these data with information on the probable nature of pollution inputs and on the assimilative capacity of the system permits prediction of potential impacts. The need for mitigation or for preventing further development is thereby identified. Public input into the assessment process may also be used directly in the objective development process.

8. Application of Water Quality Objectives

One of the most pressing problems facing water managers is how to use water quality guidelines and objectives to affect rational and scientifically defensible water quality management. The preceding seven sections have been devoted to developing a framework for the formulation and implementation of water quality guidelines and objectives; general, site-adapted, and site-specific. This information implicitly demonstrates many of the uses of these management tools.

In most situations, water quality objectives will be developed with one of two goals in mind. The first of these goals is non-degradation. The non-degradation policy may apply to systems of exceptional value, of national or provincial significance, in National Parks or other federal lands, or to pristine watercourses. The second goal is use protection, and it applies to all other watercourses.

Water quality objectives may be used as a tool with which to guide developmental activities in a drainage basin. However, to be effective, it must be possible to predict post-development water quality with the information currently at hand. Much of the information required to make these predictions is already assembled during the water quality objectives development process. Dilution models and data on effects of existing developments provide additional information required to make forecasts. Under certain circumstances, back-calculation from water quality objectives may provide a means of assessing the assimilative capacity of the system for certain classes of contaminants. This information, in turn, may be used to decide if a proposed development is compatible with the management goals for the river basin and what level of waste treatment may be required. The multiple use nature of aquatic resources (both existing and future) should be considered in the decision making process, and the possibility of re-allocation of water resources amongst users should not be ignored. In this way, water quality objectives provide a means of bridging the gap between waste and water management, and a basis for rational decision making.

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Figure 1. The Role of Water Quality Objectives in Water Quality Management.

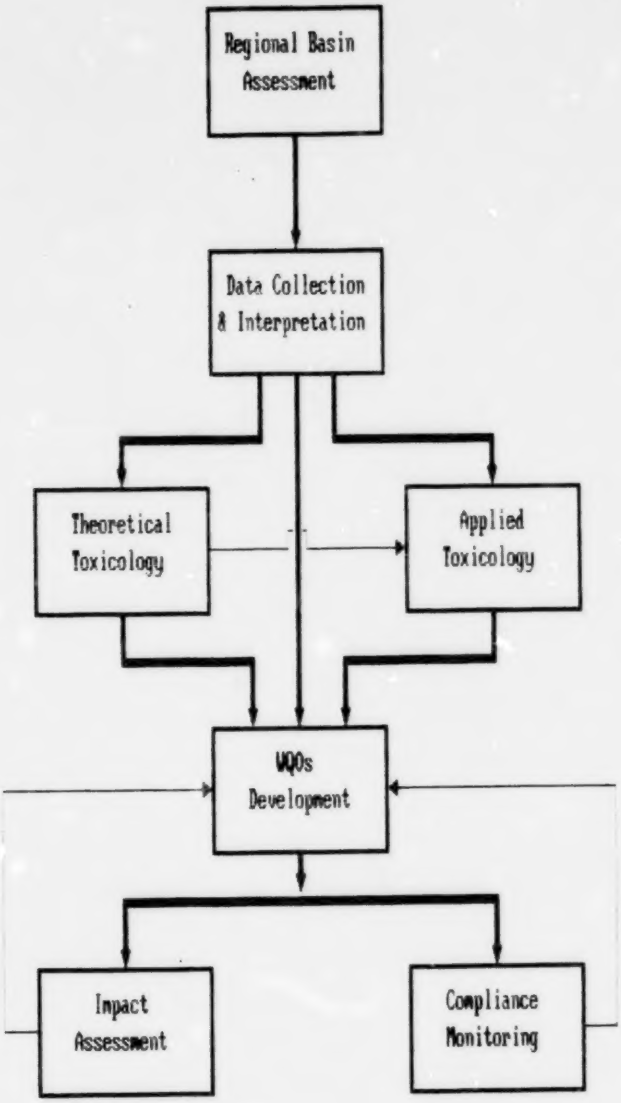


Figure 2. Regional Basin Assessment.

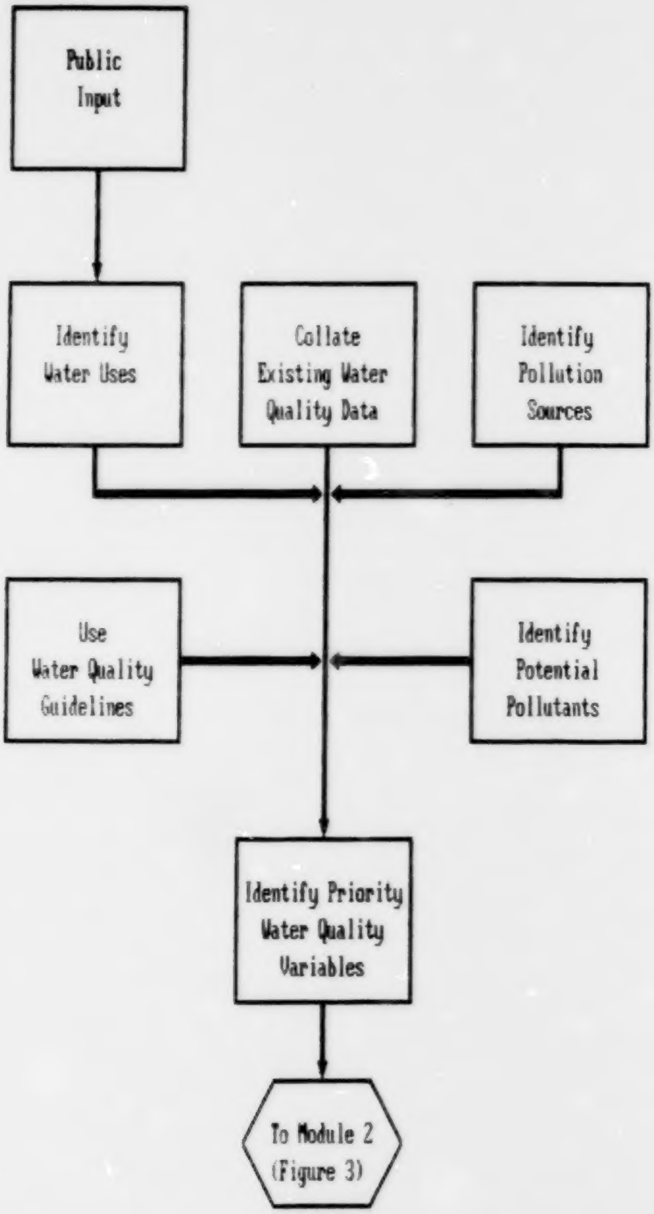

Public Input
Identify Water Uses
Collate Existing Water Quality Data
Identify Pollution Sources
Use Water Quality Guidelines
Identify Potential Pollutants
Identify Priority Water Quality Variables
To Module 2 (Figure 3)

Figure 3. Data Collection and Interpretation

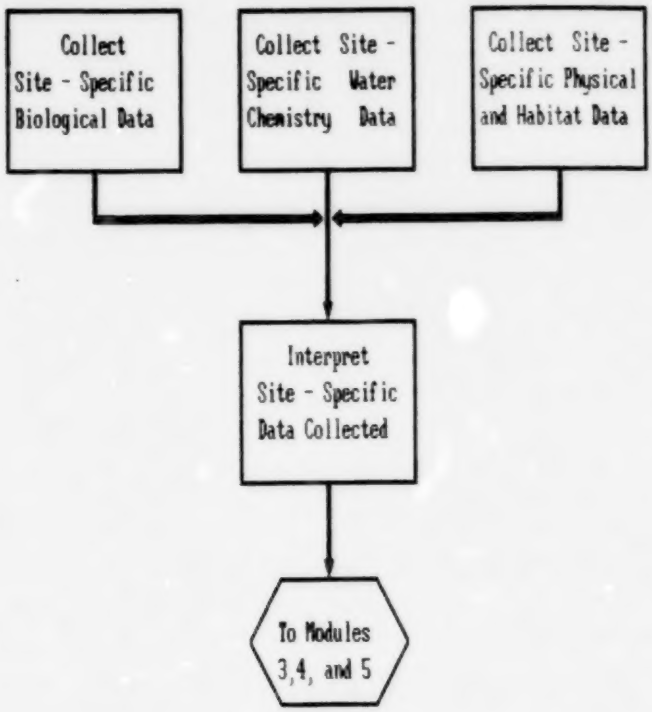

Collect Site - Specific Biological Data
Collect Site - Specific Water Chemistry Data
Collect Site - Specific Physical and Habitat Data
Interpret Site - Specific Data Collected
To Modules 3,4, and 5

Figure 4. Theoretical Toxicological Assessment

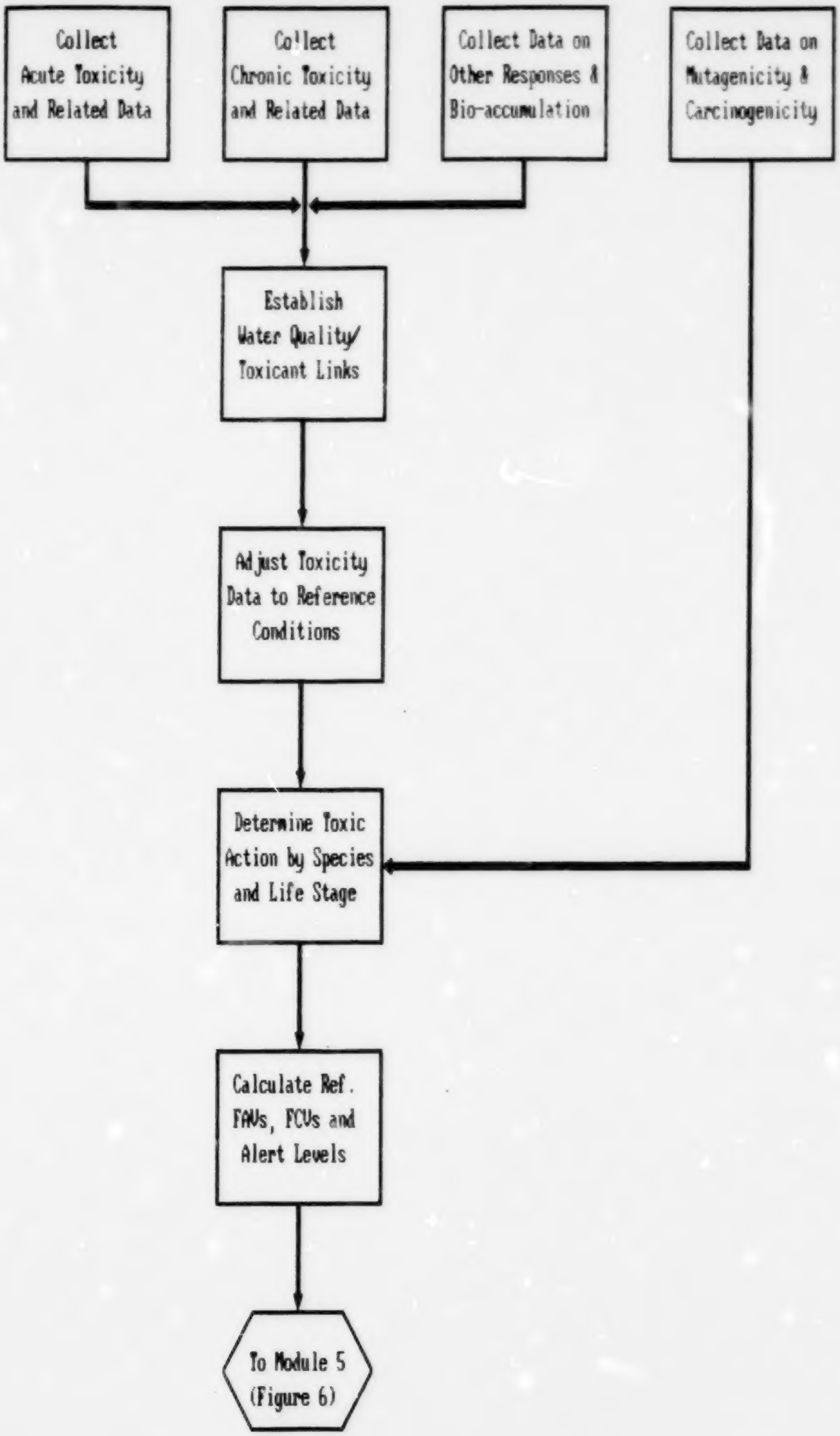
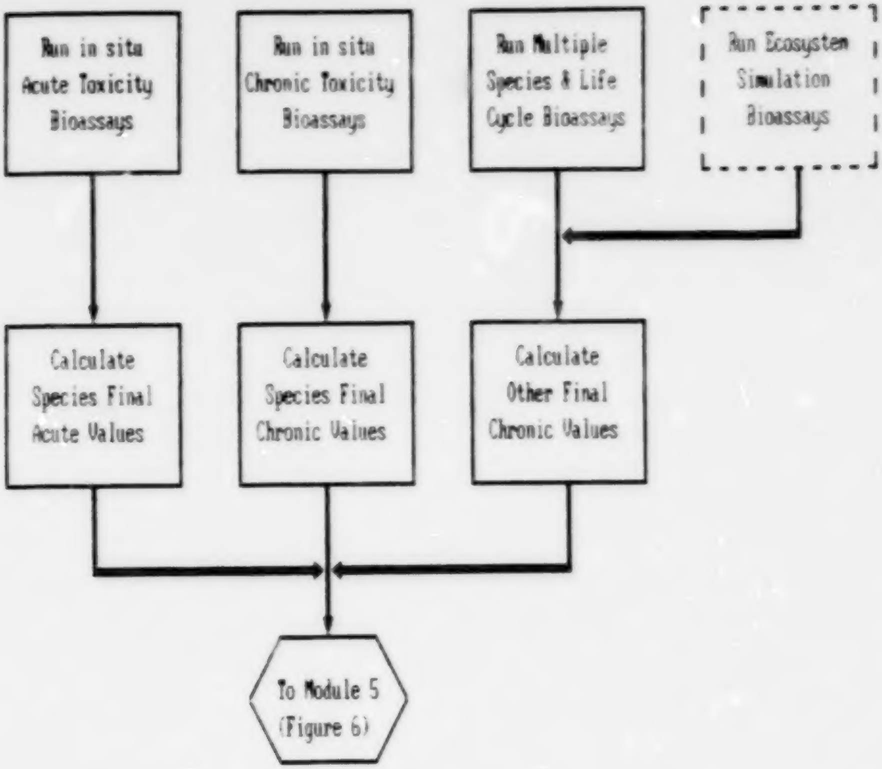

Collect Acute Toxicity and Related Data
Collect Chronic Toxicity and Related Data
Collect Data on Other Responses & Bio-accumulation
Collect Data on Mutagenicity & Carcinogenicity
Establish Water Quality/ Toxicant Links
Adjust Toxicity Data to Reference Conditions
Determine Toxic Action by Species and Life Stage
Calculate Ref. FAVs, FCVs and Alert Levels
To Module 5 (Figure 6)

Figure 5. Applied Toxicological Studies.



Run in situ Acute Toxicity Bioassays
Run in situ Chronic Toxicity Bioassays
Run Multiple Species & Life Cycle Bioassays
Run Ecosystem Simulation Bioassays
Calculate Species Final Acute Values
Calculate Species Final Chronic Values
Calculate Other Final Chronic Values
To Module 5 (Figure 6)

Figure 6. Water Quality Objectives Setting

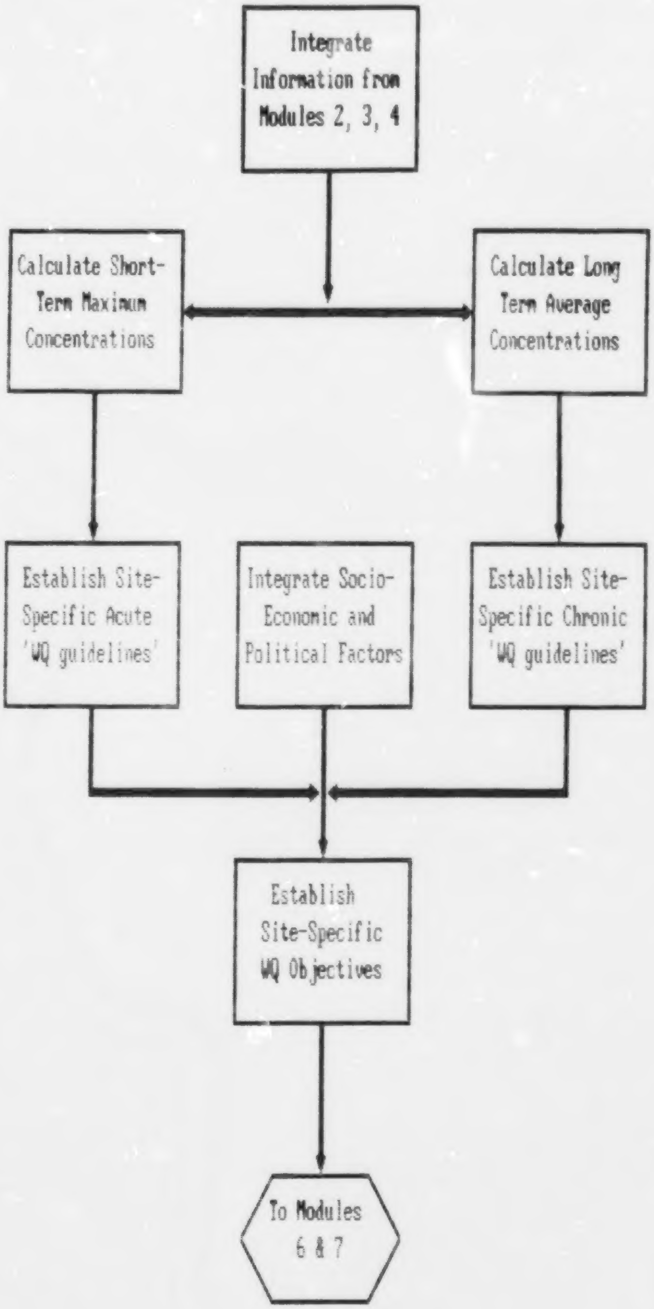

Integrate Information from Modules 2, 3, 4
Calculate Short-Term Maximum Concentrations
Calculate Long Term Average Concentrations
Establish Site-Specific Acute 'WQ guidelines'
Integrate Socio-Economic and Political Factors
Establish Site-Specific Chronic 'WQ guidelines'
Establish Site-Specific WQ Objectives
To Modules 6 & 7

Figure 7. Monitoring for Compliance with WQOs

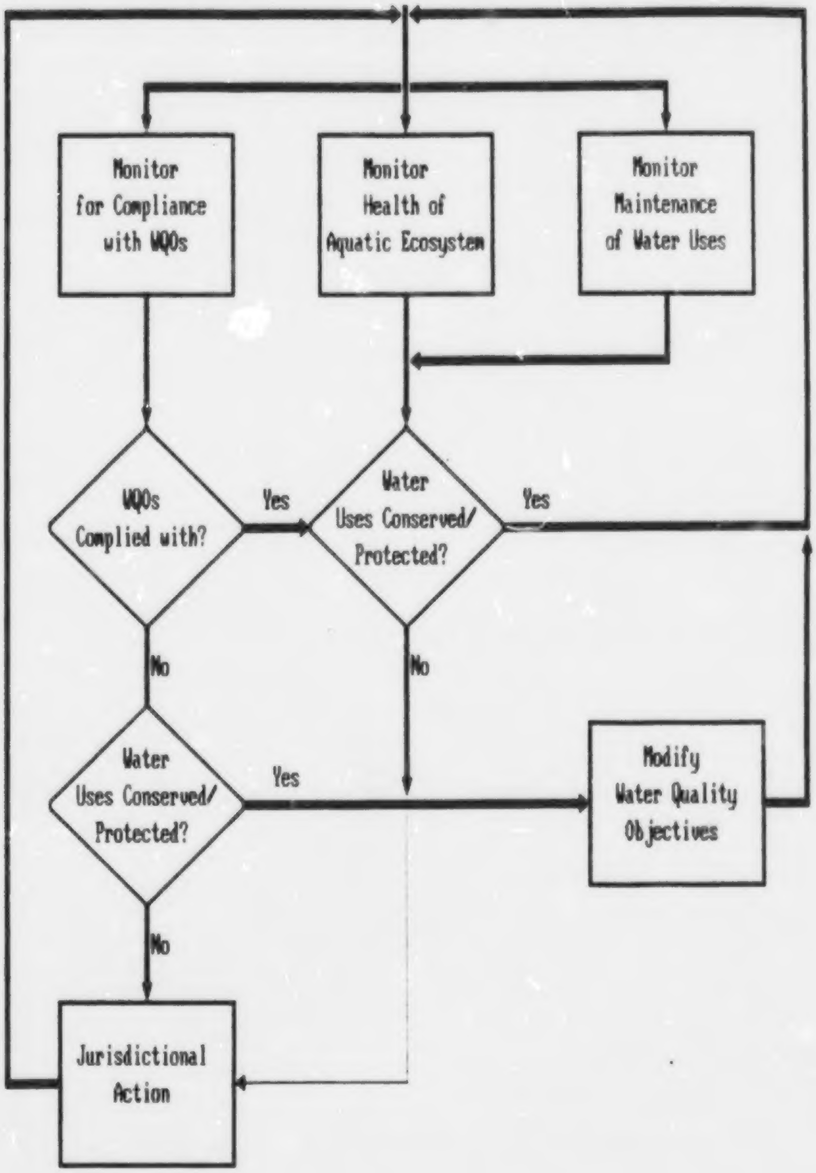
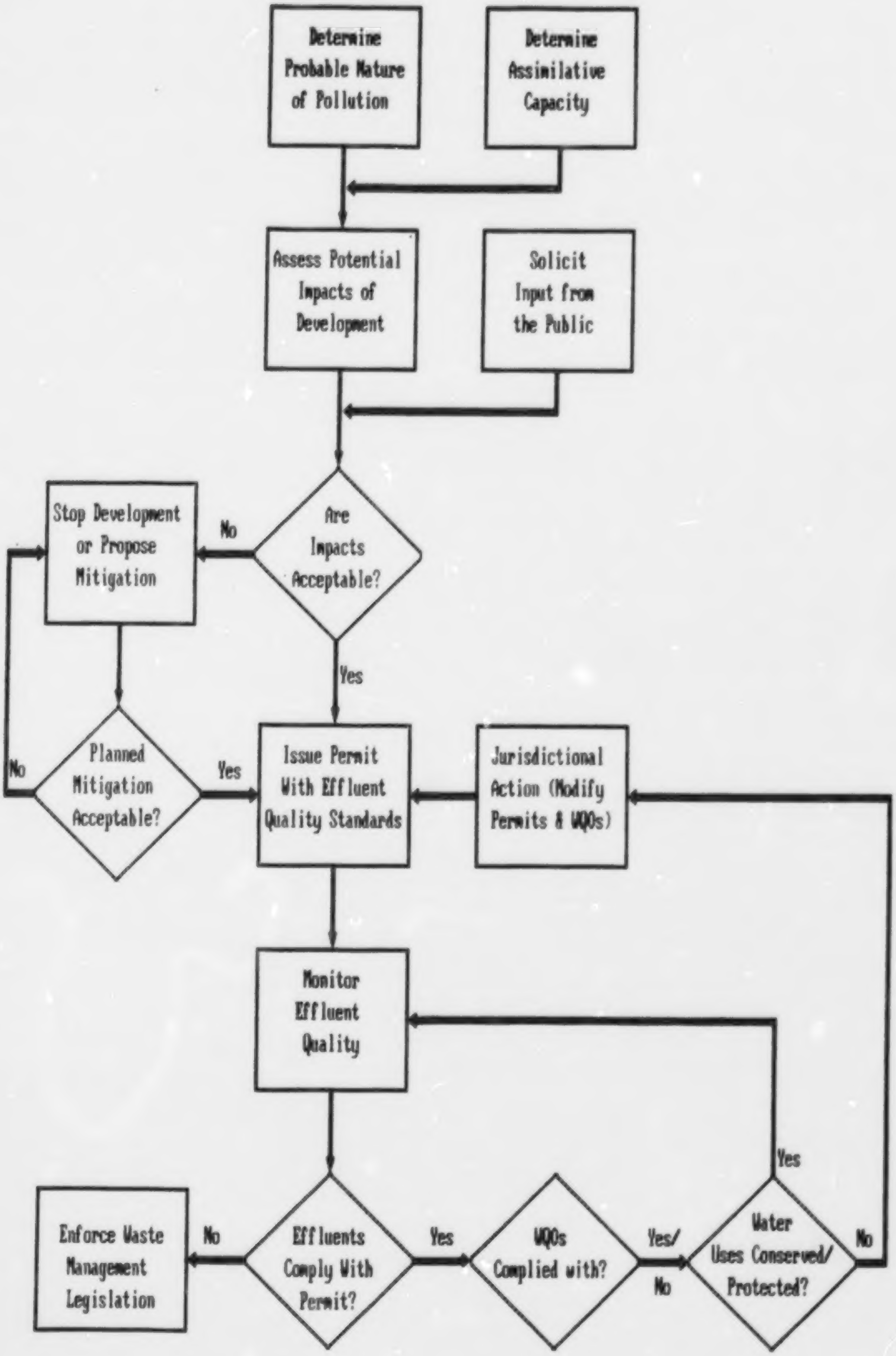

Monitor for Compliance with WQOs
Monitor Health of Aquatic Ecosystem
Monitor Maintenance of Water Uses
WQOs Complied with?
Yes
Water Uses Conserved/ Protected?
Yes
No
No
Water Uses Conserved/ Protected?
Yes
Modify Water Quality Objectives
No
Jurisdictional Action

Figure 8. Environmental Impact Assessment


Determine Probable Nature of Pollution
Determine Assimilative Capacity
Assess Potential Impacts of Development
Solicit Input from the Public
Stop Development or Propose Mitigation
No
Are Impacts Acceptable?
Yes
No
Planned Mitigation Acceptable?
Yes
Issue Permit With Effluent Quality Standards
Jurisdictional Action (Modify Permits & WQOs)
Monitor Effluent Quality
Yes
Enforce Waste Management Legislation
No
Effluents Comply With Permit?
Yes
WQOs Complied with?
Yes/
No
Water Uses Conserved/ Protected?
No

An Approach Used to Establish Site Specific Water Quality Indicators on Interprovincial Streams

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BACKGROUND

The development of water quality indicators at the interprovincial boundaries is part of the Prairie Provinces Water Board's ongoing program to promote cooperation and effective water quality management on eastward flowing interprovincial streams. The indicators (objectives) are being developed in accordance with the water quality mandate of the Prairie Provinces Water Board (PPWB) as outlined in the 1969 Master Agreement on Apportionment.

In 1973 the Board established general objectives for all eleven PPWB interprovincial monitoring sites located along the Alberta-Saskatchewan and Saskatchewan-Manitoba boundaries. These objectives were developed jointly by Canada and the Provinces of Manitoba, Saskatchewan and Alberta. At the request of the Board, Canada upgraded its interprovincial monitoring program at the 11 PPWB monitoring stations in April 1974. Routine monthly water quality sampling has been conducted at these sites since that time. In 1979, the Committee on Water Quality (COWQ) a working committee of the Board, concluded, based on the PPWB sampling program, that the 1973 PPWB Water Quality Objectives had significant limitations. The Committee agreed that no single water quality objective could be formulated for all 11 interprovincial PPWB monitoring sites that would fully meet the needs of the Board. The Board directed the COWQ to develop an approach that could be used to establish Site Specific Water Quality Indicators for each PPWB interprovincial site and to identify concentrations that should not be exceeded at the interprovincial boundary. This approach was used to ensure that the unique biological, hydrological, geochemical and demographic characteristics of the basin were adequately considered.

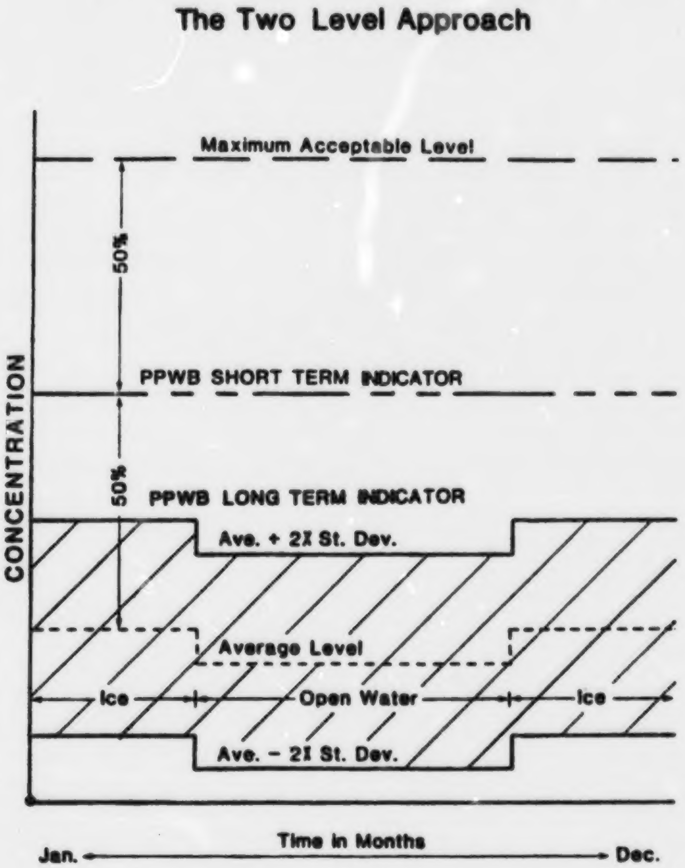
APPLICATION OF WATER QUALITY INDICATORS

These Proposed Site Specific Water Quality Indicators are a tool to identify potential interprovincial water quality concerns. They indicate acceptable in-stream water quality characteristics that, if met, should protect designated downstream uses and ensure that the resource is adequately shared by the provinces. The proposed indicators provide the Board with a basis for water quality monitoring and assessment, and detect water quality changes over the long term. They are site specific for each stream and are being developed in a consistent manner for all eleven PPWB sites. The Water Quality Indicators may be modified on the basis of changes in downstream uses or new scientific and technical information.

TWO LEVEL APPROACH FOR ESTABLISHING WATER QUALITY INDICATORS

The Committee on Water Quality has developed the Proposed Water Quality Indicators for the PPWB using a two level approach. The approach retains maximum flexibility at each individual jurisdiction while protecting the long term interests of downstream jurisdictions. The two level approach can also be applied to rivers crossing more than one boundary as it is site specific with regard to its long and short term numbers. A diagrammatic description of the two level approach is shown in Figure 1.

Figure 1. Diagrammatic Description of the Two Level Approach



The short term indicator is a concentration of a constituent that should be investigated if exceeded by an individual grab sample taken from a river at the interprovincial boundary crossing. The exceedance of a short term indicator identifies an immediate water quality concern. The short term indicator is established halfway between the annual mean or seasonal mean concentration of the stream and the maximum acceptable level. Short term water quality indicators reflect a sharing of the resource and permit the upstream province to develop within reasonable and acceptable levels while protecting similar development in the downstream province.

The long term indicator is a range of a constituent that should be investigated if exceeded by a seasonal or annual mean concentration from a river at the interprovincial boundary crossing. The exceedance of a long term indicator identifies a possible long term water quality concern. A long term water quality indicator is generally based on two standard deviations from the 1974-1982 historical annual or seasonal mean concentration of the river. A long term water quality indicator is developed to protect the integrity of the stream. It alerts the Board to long term trends in water quality before short term indicators are exceeded.

PROCESS OF FORMULATING WATER QUALITY INDICATORS

Four main steps are carried out in the process of developing the site specific indicators. The first step in the process is to have the downstream province identify the present and future water uses it wishes to protect. Parameters of concern are also identified for the protection of each designated downstream use. A parameter use matrix is then prepared. This matrix identifies constituents that must be limited to ensure the protection of each specific use. Uses identified on this matrix typically include domestic consumption, aquatic life and wildlife, industrial consumption, irrigation consumption, livestock consumption and recreation. Table 1 shows one use (industrial consumption) extracted from the total use matrix. The parameters listed, such as, total alkalinity, ammonia, chloride, etc. are critical for the protection of industrial use. A maximum acceptable level that should not be exceeded for each parameter is then determined to ensure the protection of the most sensitive of the designated uses. This level is determined from the most recent criteria documents and scientific literature. If criteria documents are not available for some constituents (i.e. major ions) professional judgement is used and a number is negotiated by the COWQ.

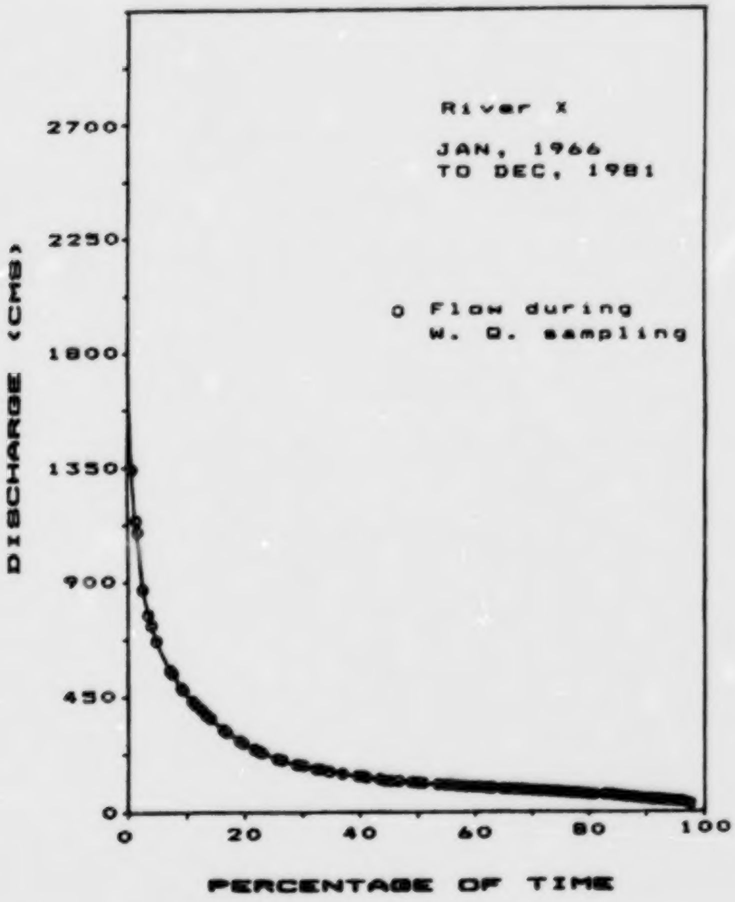
Table 1. Industrial Consumption Parameters from the Total Use Matrix

Recreation	Industrial Consumption	Domestic Consumption
	Acidity	
	Alkalinity (Total)	
	Alkalinity (Bicarbonate)	
	Aluminum	
	Chloride	
	Colour	
	Filterable Residue	
	Hardness	
	Iron	
	Manganese	
	Non-Filterable Residue	
	pH	
	Phosphate	
	Silica	
	Sulphate	

The second step is to review the relevant historical data base for that stream, verify the results of the nine years of data (1974 - 1982) and to remove spurious data by using acceptable statistical practices and consulting of the provincial and federal data.

The third step, is to compare water quality sample coverage with historic streamflow to ensure that water quality samples have been collected throughout the entire range of flow. The flow at the time each sample is collected is plotted on a flow duration curve, as shown in Figure 2, to illustrate the water quality sample coverage. Water quality samples for this stream were collected throughout the entire discharge range. Once it has been determined that water quality samples are representative of all flow conditions at that site, then a detailed statistical analysis is meaningful.

Figure 2. Water Quality Sample Coverage for River X



The fourth step in developing indicators is to conduct a detailed statistical analysis of the data base. Since some constituents fluctuate considerably between seasons, the Committee expressed concern with establishing an annual indicator that would protect the stream during one season but would allow a large increase at other times of the year. The water quality data base was, therefore, split into two separate seasons, ice cover and open water. If the ice cover and open water data for a particular parameter are determined to be statistically different, based on the population size, the normality and the distribution, a seasonal water quality indicator is established. If there is no statistical difference between the two seasonal populations, then an annual water quality indicator is established. Detailed seasonal and annual statistics for each parameter are run. An example of the statistical determinations for sodium dissolved in river X is shown in Table 2.

Table 2. Sodium Statistics for River X

RIVER X WATER QUALITY STATISTICS

Sodium (diss.)

Statistics	Annual	Ice Cover	Open Water
No. of Samples	95	43	52
Mean	16.5	18.2	15.1
Median	17.0	18.0	15.3
Variance	9.72	6.73	7.83
Maximum	24	24	22
Minimum	10	12	10
Range	14	12	12
Skewness	0.0055	-0.11	0.25
1 Std. Dev.	3.12	2.59	2.8
2 Std. Dev.	6.24	5.18	5.6
Flow	22-1340	22-450	30-1340
90th Percentile	20.5		

After the preliminary work has been completed, Site Specific Water Quality Indicators, annual and seasonal where applicable, can be determined for each parameter. For those parameters that a maximum acceptable concentration can be determined, from the criteria documentation and scientific literature, short term indicators are developed based on a level halfway between the maximum acceptable and the mean historical concentrations. Long term indicators for these parameters are based on the historical annual or seasonal mean concentrations plus two standard deviations. An example application of this general two level approach is shown in Table 3.

EXCEPTIONS TO THE TWO LEVEL APPROACH

The two level approach is applied as a rule wherever possible but modifications were necessary for some constituents. The Committee on Water Quality, after considerable negotiations, agreed upon a number of concepts or rules for these modifications that have been applied consistently so that parameters with similar characteristics are always treated the same.

Slight modifications to the general approach were necessary for establishing water quality indicators for some specific groups of constituents. Major ions were treated differently since appropriate maximum acceptable levels for major ions are not available in the scientific literature. The short term indicators for major ions are established using the 90th percentile plus 50%. The 50% increase was agreed upon by the Committee to allow for future development in the basin but still protect aquatic communities in the river. The long term indicators for major ions are based on two standard deviations from the historical, annual or seasonal mean concentrations of the river. An example of the major ion approach for establishing short term and long term indicators is shown in Table 4.

Table 3. Application of the Two Level Approach for Arsenic

ARSENIC - DISS	
Mean Quality in River X	0.00075 mg/L
Most Sensitive Use	Drinking Water
Max. Accept	0.05 mg/L Canadian DW Guidelines
Short Term Indicator	$\frac{\text{Max. Accept + Hist. X Conc.}}{2}$ 0.025 mg/L Annual
Long Term Indicator	Mean +/- 2 Stand. Dev. 0.002 mg/L Annual

Table 4. Application of the Major Ion Approach for Calcium

CALCIUM	
Mean Quality of River X	52 mg/L
Most Sensitive Use	Aquatic
Max. Acceptable Conc.	None in literature
Short Term Indicator	90th percentile + 50% 77 mg/L Annual
Long Term Indicator	Mean +/- 2 Stand. Dev. Ice Cover 40-57 mg/L Seasonal Open Water 33-44 mg/L

The nutrient group was also treated differently since they are known to have increased substantially by man's activities in the prairies. The COWQ agreed that total phosphorus is a long term concern and that a short term indicator is not necessary. Since total phosphorus concentrations in most prairie streams are relatively high and near maximum acceptable levels, due to both natural and municipal inputs, the Committee felt that it would not be appropriate to allow a two standard deviation increase over the streams historical mean concentration. The COWQ agreed to estimate the background total phosphorus concentration for some streams and establish a long term indicator, based on a level halfway between the maximum acceptable level and the estimated background concentration. The estimated natural concentration was determined from a review of all water quality data available for the basin including federal and provincial routine monitoring data as well as special study data. An application of this approach is shown in Table 5.

The long term indicators for nitrogen forms are based on the historical mean concentration plus one standard deviation. One standard deviation is used because nitrogen may be a potential contributing factor to nuisance aquatic organisms in some interprovincial rivers. The ammonia short term indicators are established on acute toxicity criteria and are therefore dependent on the pH and temperature of the stream.

Table 5. Application of the Nutrient Approach for Total Phosphorous

PHOSPHOROUS TOTAL	
Mean Quality of River X	0.21 mg/L
Estimated Background Conc.	0.095 mg/L
Most Sensitive Use	Aquatic Weed Growth
Max. Acceptable Conc.	0.1 mg/L
Short Term Indicator	None
Long Term Indicator	$\frac{\text{Max. Acceptable + Estimated Natural}}{2}$ 0.1 mg/L Annual

A special approach was necessary for biological indicators because of their exponential growth patterns. The short term indicators for total coliform and fecal coliform are based on the provincial surface water quality objective and the Guidelines for Canadian Recreational Water Quality respectively. The long term indicators for coliforms are based on the geometric mean plus one geometric standard deviation so as to better represent coliform growth patterns.

Since organic insecticides are normally not detected in interprovincial waters a special approach was developed to deal with these parameters. The Committee felt that interprovincial rivers should be free of substances in concentrations or combinations that accumulate in the environment and are toxic or may be harmful to human, animal or aquatic life. The Committee agreed not to set specific water quality indicators for organic insecticides but rather to make a general statement that these substances should not be present in streams crossing the interprovincial boundary.

PUTTING THE WATER QUALITY INDICATORS TO WORK

If the Proposed Site Specific Water Quality Indicators are being used as a working tool for the Board, the Committee on Water Quality will inform the Board if any short term or long term indicators are exceeded. The excursion of a short term indicator will be assessed on a single grab sample while the excursion of a long term indicator will be based on a seasonal or annual mean. The Committee will review the excursions, carry out additional grab sampling or special studies to investigate the situation, supply an explanation of these excursions, and suggest a recommended course of action to the Board. Jurisdictions will also be responsible for raising any problems caused by these excursions. The Board will then make a final decision and recommend what action is required to resolve the situation.

RATIFICATION OF WATER QUALITY INDICATORS

The Committee on Water Quality has proposed Site Specific Water Quality Indicators at six of the eleven PPWB monitoring sites and will continue to develop these indicators on the remaining five PPWB sites. The Board has not approved these indicators but is in the process of determining the equity of this approach. It is anticipated that these indicators will be tested for a trial period at all PPWB sites. The indicators may be modified or refined as a result of provincial agency review. Once the review and test process is complete, the Board may approve these Water Quality Indicators and recommend their adoption by the Federal and the three Provincial Governments. These indicators would then be a tool which could be used to prevent future interprovincial water quality problems from occurring.

REFERENCES

Environment Saskatchewan. January 1975. Surface Water Quality Objectives, in: "Water Quality Objectives."

Health and Welfare Canada. October 1983. "Guidelines For Canadian Recreational Water Quality," Canadian Government Publishing Centre, Ottawa.

Prairie Provinces Water Board. October 1969. "Master Agreement on Apportionment."

Introducing Cesars

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M4V 1P5

INTRODUCTION

Cesars (Chemical Evaluation Search and Retrieval System) is a data base containing information on chemicals of environmental concern. CESARS was originally developed in 1979 by the Michigan Department of Natural Resources with funding from U.S. EPA. The CESARS information was used to assess chemicals for placement on Michigan's Critical Materials Registry (CMR).

In 1985, the Ontario Ministry of the Environment initiated work on the "Vector Scoring System" to be used to assess and priorize chemicals of concern in Ontario. Ontario based their information gathering and summarization of the scoring system used for the CMR. The Ontario system was further modified to meet the needs of the MISA program, especially the preliminary assessment process which Ontario developed specifically for the MISA program.

In 1989, Ontario and Michigan signed a Memorandum of Understanding with the Canadian Centre for Occupational Health and Safety (CCOHS) to distribute CESARS and make it available at a reasonable price. CCOHS provided the system expertise and capability to make the database available on-line across Canada and available on CD ROM around the world.

The memorandum also outlined that Michigan and Ontario had agreed on the protocols for the detailed collection and summarization of chemical information. In addition, the assessments for agreed upon chemicals would be contained in the CESARS database and both jurisdictions would maintain a high level of quality control for the database. Further, the preliminary assessment process developed by Ontario using the Vector Scoring System would be available as a topic area in CESARS.

To date, CESARS contains about 400 detailed assessment and 600 preliminary assessments. About 140 detailed and 240 preliminary assessments are currently available on CCINFO. That number will increase substantially with each 3 month update of the database. A list of chemicals assessed and those available through CCINFO can be obtained directly from the author.

An overview of the process for developing detailed and preliminary assessments is discussed below:

1. List of Pollutants

To initiate the process of producing CESARS detailed and preliminary assessments, a list of pollutants is required. The list is usually derived from scans of effluents, air, water, soil, sediments, etc. or from the literature. The chemicals identified on the scans must be verified and the appropriate CAS numbers must be assigned. Complete lists of synonyms need to be collected as well.

2. Literature Search

Both preliminary and detailed assessments involve a comprehensive search of the literature for available information. For detailed assessments the search includes more topic areas and use of the primary literature. Further details of the information sources and the literature search strategy are discussed later.

3. Scoring System

Each chemical is assigned a numerical score based on their environmental behaviour, exposure potential, and adverse effects on human, animal and plant life.

4. Topic Areas

The information in CESARS is separated into the following 23 topic areas:

TOPIC AREA DESCRIPTIONS

1) Properties

This topic area contains information on the physical-chemical properties of a chemical such as boiling point, odour, vapour pressure, octanol/water partition coefficient, etc.

2) Regulation

This topic area contains information on regulations and guidelines in the United States and Canada.

3) Manufacture

This topic area contains a short description of the uses, occurrence, production and methods of synthesis for the chemical.

4) Acute Toxicity: Terrestrial Animals

This topic area contains information on the effects of acute exposure to experimental animals. The principal routes of exposure considered are oral, inhalation and dermal. An exposure is considered acute if the animal received one dose only or multiple doses in a 24 hour period only. The most common acute exposure tests are LD_{50} and LC_{50} tests which measure the median lethal dose or concentration.

5) Acute Toxicity: Humans

This topic area contains information on the effects of a chemical to humans after one exposure or multiple exposures within 24 hours.

6) Acute Toxicity: Aquatic Animals

This topic area contains information on acute toxicity studies on aquatic animals. Emphasis is given to data on freshwater species and studies giving median lethal or effective concentrations (LC_{50} or EC_{50}).

7) Chronic Toxicity: Terrestrial Animals

This topic area contains information from subchronic (subacute) and chronic toxicity studies on terrestrial animals. If available the No-Observed-Effect-Level (NOEL) is reported. Carcinogenicity, Reproductive, Teratogenic, or Genotoxic effects are reported elsewhere in the appropriate topic areas.

8) Chronic Toxicity: Humans

This topic area reports information from studies on the subchronic (subacute) and chronic effects of chemicals to humans. If available the No-Observed-Adverse-Effect-Level (NOAEL) is reported. Carcinogenic, Reproductive, Teratogenic, or Genotoxic effects are reported elsewhere in the appropriate topic areas.

9) Chronic Toxicity: Aquatic Animals

This topic area contains information about subchronic (subacute) and chronic toxicity to aquatic life. Emphasis is given to data on freshwater fish and marine macroinvertebrates. The Maximum Acceptable Toxic Concentration (MATC), the Highest-No-Observed-Adverse-Effect-Level (HNOAEL) and the Lowest-Observed-Adverse-Effect-Level (LOAEL) are given when available.

10) Phytotoxicity

This topic area contains information on the effects of substances on aquatic and terrestrial plants.

11) Carcinogenicity

This topic area contains the results of animal and human studies designed to investigate the potential of a chemical to cause cancer. The IARC (International Agency for Research on Cancer), NTP (U.S. National Toxicology Program) and NIOSH (National Institute for Occupational Safety and Health) determinations are included.

12) Mutagenicity

This topic area contains information on various types of mutagenicity studies which utilize in vivo and in vitro techniques for evaluating the mutagenic potential of a chemical. The studies are grouped according to the following categories:

Gene mutation
Chromosomal aberration
DNA damage
Other studies

13) Reproductive and Developmental Effects: Terrestrial Animals

This topic area contains information on reproductive parameters such as parental reproductive function and activity, fertility, conception, pregnancy, and the growth and development of the offspring from conception to maturity. Reproductive effects to aquatic animals are reported in the aquatic animal topic areas. Studies on plants are in the phytotoxicity topic area.

14) Other Adverse Effects

This topic area contains information on aesthetic effects and other adverse effects for which no specific fields are available within the other topic areas.

15) Pharmacokinetics/Metabolism

This topic area contains summary information on the uptake, distribution, biotransformation and elimination of a compound by terrestrial animals.

16) Bioaccumulation/Bioconcentration

This topic area contains information on the bio-uptake (concentration and accumulation) of chemicals by freshwater and marine organisms.

17) Transport Processes

This topic area contains information from studies investigating sorption and volatilization.

18) Environmental Fate Process, General

This topic area contains information on microcosm studies, field studies, microbial effects studies and other fate processes for which no specific field exists.

19) Transformation Processes

This topic area includes information from biodegradation, reduction/oxidation, hydrolysis, photolysis and photo-oxidation studies.

20) Analysis and Treatment

This topic area contains brief information on standard analytical methods, drinking water treatment and waste/wastewater treatment.

21) References

This topic area contains the references that are cited in topic areas 1 to 20.

22) Summary

This topic area contains only the information from the summary fields of the CESARS record.

23) Ontario Environmental Assessment

This topic area contains the assessment and scoring of chemicals, as prepared by the Ontario Ministry of the Environment. The Preliminary Assessment is prepared from a quick review of secondary literature sources; the Detailed Assessment is from the information in a CESARS record.

More information on topic areas is given later.

5. QA/QC

Michigan and Ontario have developed a comprehensive, multistep quality control/quality assurance process to maintain a high level of integrity for the CESARS database.

QA/QC Procedures

After the information has been extracted from the literature summarized under the appropriate topic areas or fields, a machine readable file is generated. At this point, the Quality Assurance Program is applied. This program consists of a two step process:

1. The computerized versions are reviewed for any errors by direct comparison with the original sources of information. The chemical reviews are assigned to mammalian toxicologists, aquatic biologists, and properties/environmental fate specialists for review. In this review phase, the technical content of the evaluation is considered along with format and style. Each study is reviewed for placement into the correct fields, for clear and concise language, for proper format, and for presence of all pertinent information so that consistency is maintained throughout the entire evaluation. In particular, numbers and units are inspected since these represent a critical portion of the chemical evaluation.

2. The final draft copy of CESARS information is reviewed by an information specialist for consistency with the other evaluations in the System. This final review checks to see that all the required

fields have been filled in, such as the CAS search date field, the Chemical Abstract volumes searched fields, etc. The format is looked at closely for all fields to insure that numerical fields contain only numbers, that all data is referenced, and that the references are properly cited. A number of additional details are checked and any additions or modifications as noted on the final draft are entered into the computer system.

6. Uses of Preliminary Assessments

The preliminary assessments may be used for:

1. Prioritizing lists of pollutants.
2. Developing monitoring programs.
3. Conducting rapid hazard assessments of chemical pollution problems.
4. Identifying data gaps.
5. Identifying and prioritizing multi-media contaminants.

7. Uses of Detailed Assessments

Detailed assessments may be used for:

1. Developing standards/objectives/guidelines for the control of hazardous substances in all media (air, water, waste, soil, sediment, and food).
2. Listing/delisting of toxic chemicals for various applications such as identifying hazardous wastes.
3. Assessing chemical hazards for decommissioning landfill sites.
4. Assessing chemical hazards for environmental approvals and discharge permits.
5. Identifying areas needing further environmental and toxicological research.
6. Providing the public with information on chemicals of health and environmental concern.

INFORMATION SOURCES AND LITERATURE SEARCH STRATEGY

Information Sources for Preliminary Assessments

The data contained in preliminary assessments are derived mainly from secondary literature sources. Several on-line databases and standard reference books are routinely used to access information (see Figure and Table attached). This is referred to as a Level I search. However, when sufficient information is not found for a particular parameter, a Level II search is undertaken. This involves using abstracts from additional on-line databases. Also, primary references are occasionally sought.

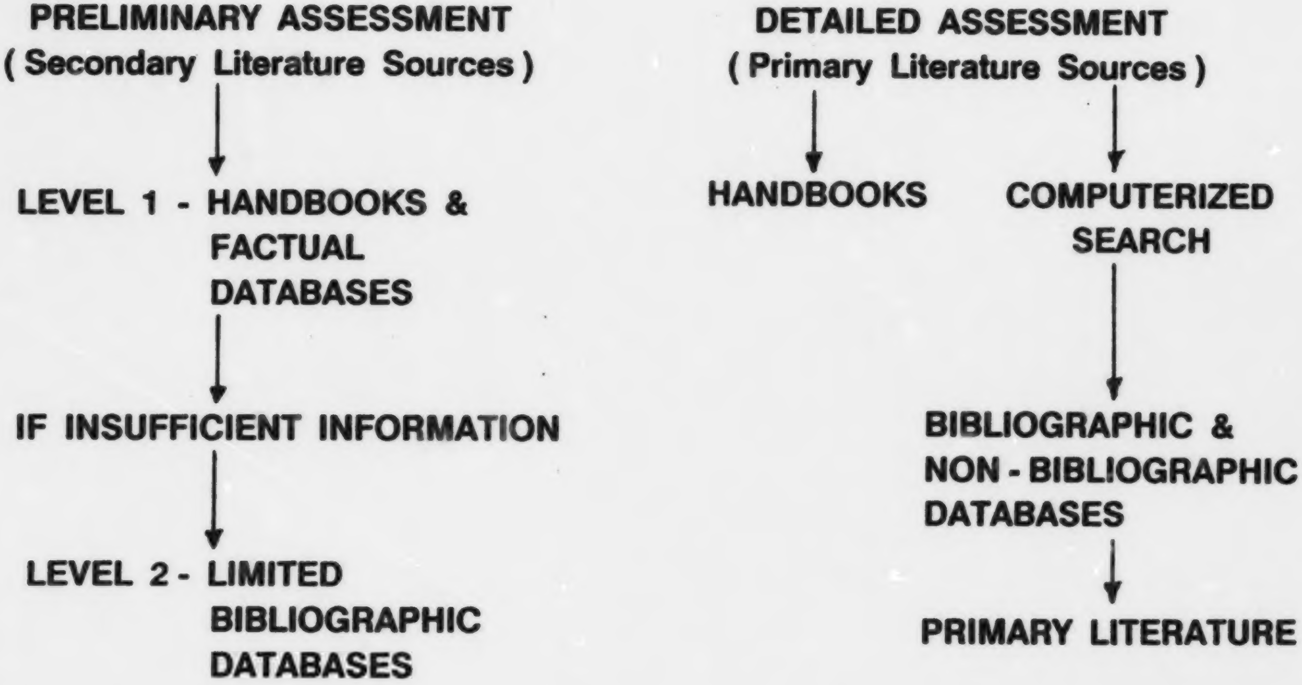
Information Sources for Detailed Assessments

The data contained in CESARS are derived mainly from primary literature sources. Several on-line databases and standard reference books are routinely used. In addition to the information sources used for Preliminary Assessments, other databases such as TOXLIT, BIOLOG, DATALOG, CHEMFATE, TSCATS, IRPTC, and TSCAPP are searched.

After a comprehensive literature search is completed, references are identified, collected and reviewed for inclusion into all the CESARS fields. Secondary information sources are acceptable for the 'properties', 'regulation', and 'manufacture' sections.

In some cases, when the primary literature database is overwhelming, literature reviews may be used. Reviews from accepted world authorities are referenced as source documents if no other information is available. In addition, the reviewer will pursue other studies to update and expand the available database.

LITERATURE SEARCH STRATEGY



PRELIMINARY ASSESSMENT
(Secondary Literature Sources)

LEVEL 1 - HANDBOOKS &
FACTUAL
DATABASES

IF INSUFFICIENT INFORMATION

LEVEL 2 - LIMITED
BIBLIOGRAPHIC
DATABASES

DETAILED ASSESSMENT
(Primary Literature Sources)

HANDBOOKS

COMPUTERIZED
SEARCH

BIBLIOGRAPHIC &
NON - BIBLIOGRAPHIC
DATABASES

PRIMARY LITERATURE

Information Sources for Preliminary Assessments

Level I Search Sources (Printed Texts)

American Conference of Governmental Industrial Hygienists (ACGIH), 1986 - Documentation of Threshold Limit Values for Substances in Workroom Air
Center for Lake Superior Environmental Studies (CLSES), 1987 - Acute Toxicities of Organic Chemicals to Fathead Minnows
Clayton and Clayton, 1982. Patty's Industrial Hygiene and Toxicology
Hawley, 1981 - The Condensed Chemical Dictionary
Hayes, 1982 - Pesticides Studied in Man
International Agency for Research on Cancer (IARC), 1972 - IARC Monographs series
Mayer and Ellersieck, 1986 - Manual of Acute Toxicity
Michigan Dept. of Natural Resources - CESARS Chemical Profiles and CMR Hazard Assessment Profiles
Packer, 1975 - Nanogen Index
Sax, 1984 - Dangerous Properties of Industrial Materials
Shepard, 1986 - Catalog of Teratogenic Agents
Sittig, 1985 - Handbook of Toxic and Hazardous Chemicals and Carcinogens
Soderman, 1982 - CRC Handbook of Identified Carcinogens and Noncarcinogens: Carcinogenicity / Mutagenicity Database, Vol. 1
U.S. National Research Council, 1977-87 - Drinking Water and Health, Vols. 1-7
U.S. National Toxicology Program (NTP), 1980-86 - Bioassay for Possible Carcinogenicity
Verschueren, 1983 - Handbook of Environmental Data on Organic Chemicals
Windholz et al., 1983 - The Merck Index
Worthing, 1987 - The Pesticide Manual
Other reviews from recognized agencies

Level I Search Sources (On-Line Databases)

AQUIRE (aquatic toxicity and bioaccumulation)
BIODEG (environmental fate)
CA Registry (synonyms, old CAS numbers)
CCRIS (carcinogenicity)
CHEMFATE (chemical properties, bioaccumulation factors, environmental fate)
ENVIROFATE (environmental fate)
GENETOX (genotoxicity and mutagenicity)
HSDB (chemical properties, environmental fate, toxicity)
ISHOW (chemical properties)
LOGP (partition coefficients)
PHYTOTOX (phytotoxicity)
RTECS (toxicity)

Level II Search Sources (On-Line Databases)

CA Search (all fields)
NTIS (environmental fate)
QSAR (estimates of chemical properties, biodegradation, toxicity)
TOXLINE (acute and sublethal toxicity)

Setting Provincial Water Quality Guidelines

Conrad de Barros
Environment Ontario
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INTRODUCTION

Provincial Water Quality Guidelines (PWQGs) are a set of substance-specific numerical criteria developed primarily for the protection of aquatic life. The clear intention of the guideline development process is to make the best use of a limited information base (i.e. an information base that is not sufficiently complete to set a provincial water quality objective), in order to protect all forms of aquatic life at all stages in an aquatic life cycle during indefinite exposure. A Guideline value is calculated by applying a safety factor (in this procedure it is called a "final uncertainty factor") to the lowest water concentration shown to cause a deleterious biological effect. The size of the safety factor itself is based on a substance's physical-chemical properties and the comprehensiveness of the aquatic toxicity data base.

Guidelines are used, like Provincial Water Quality Objectives (PWQO), to interpret data on the concentration of substances in effluents and receiving water. If ambient levels exceed the Guideline, a water pollution problem may exist. Thus Guidelines help the Ministry identify priorities for management decisions and action, and provide dischargers and abatement managers with a planning tool.

As new data become available, Guidelines will be reassessed and, if warranted, modified using the Ministry's PWQO setting process. The Guideline setting process, itself, requires identification of the scientific research needed to support this step.

The development of a Provincial Water Quality Guideline follows a prescribed series of steps, that will help ensure consistency in the interpretation and use of data for setting numerical values. Each step is described in detail in Section 3.0 of the report entitled "Ontario Water Quality Objective Development Process".

In overview, the process is as follows:

- Critical evaluation of all available data on physical-chemical properties and aquatic toxicity, obtained from a review of the world literature;
- Selection of the baseline uncertainty factor based on octanol-water partition coefficient (K_{ow}) for organics and organo-metals;

- Calculation of the final uncertainty factor based on the quantity and quality of the different types of available toxicity information and the baseline uncertainty factor. Application of the final uncertainty factor to the lowest water concentration associated with an adverse biological effect, to derive a preliminary Guideline.

- Assessment of data on other adverse effects such as organolepsis (taste and odour of water and tainting of fish tissue) and bioaccumulation and derivation of preliminary Guideline(s).

  All information and data employed in the derivation process are documented in a short report, prepared in a prescribed format. The preliminary Guidelines based on aquatic toxicity, organoleptic effects, and bioaccumulation are compared and the one which is the most stringent (lowest) is recommended as the PWQG.

- Final review by MOE Aquatic Criteria Develoment Committee. This group, which maintains an overview of all PWQO and Guideline setting activities, may make minor adjustments to the Guideline value in light of information used in its derivation. Any such changes would be fully rationalized and reported.

REFERENCE

Ontario's Water Quality Objective Development Process. May 1989. Aquatic Criteria Development Committee, Environment Ontario. Draft Document.

RATIONALIZATION OF LIQUID WASTE OBJECTIVES: TOXIC SUBSTANCES

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Environment Quebec
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Ste-Foy, Quebec
G1X 4E4

This paper is taken directly from the visual presentation

NOTICE

1. Since the regulation respecting certification has been tabled, we felt it would be advisable to modify the transparencies presenting our procedure in order to adapt it to this future reality. Therefore, the terms "attestation requirements" and "attestation" are used loosely until we know the terminology that will be adopted by the administration.

2. Figure 3 presents the position of the Aquatic Environment Quality Branch (DQMA). Since discussions of this point have not yet been initiated, it does not presently represent MENVIQ's position.

Mandate of the DQMA

1. To identify constraints to the protection of human health and biological resources.

2. To identify these constraints with a view to the maintenance and recovery of the uses of water and of the aquatic and terrestrial biological resources that depend on it.

3. To develop environmental release objectives for specific point sources of water pollution.

Objective of the presentation

To describe the main steps in establishing environmental release objectives, and the tools developed by DQMA for each step for the protection of public health and aquatic life.

Characteristics of the technique

1. environmentally safe

2. allows for the possibility of making quantitative recommendations within the framework of Quebec's water treatment program

3. sufficiently rapid to keep up with the water treatment program

4. reveals problematic cases for which additional research is necessary.

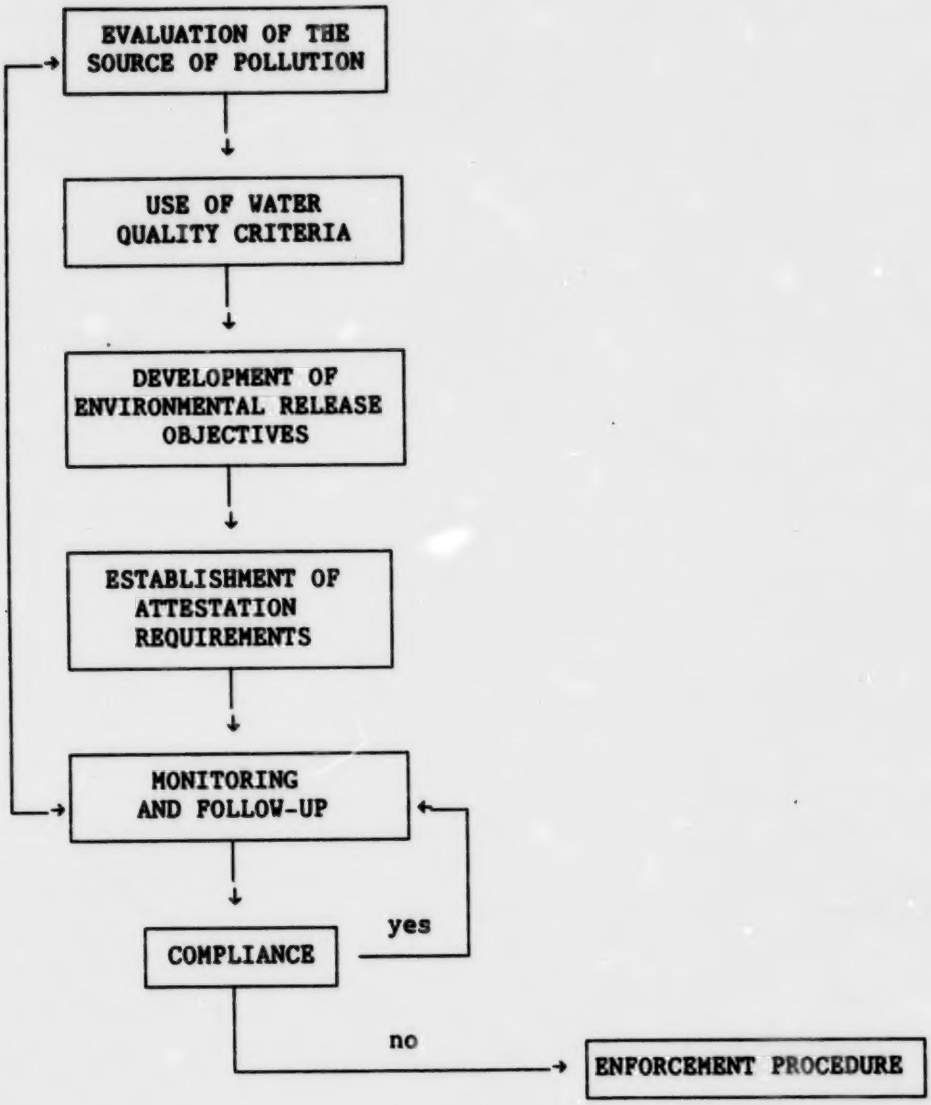
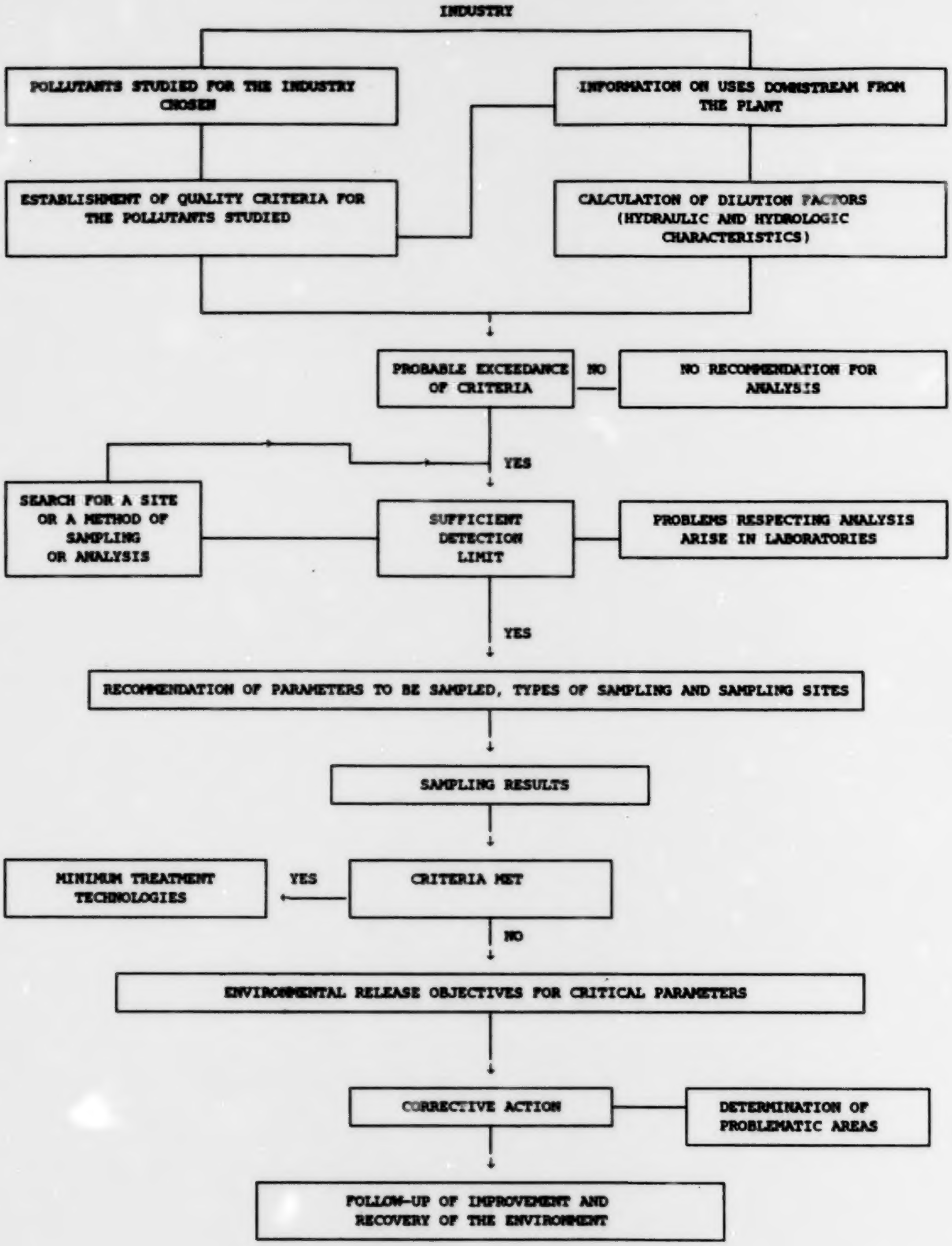

EVALUATION OF THE SOURCE OF POLLUTION
USE OF WATER QUALITY CRITERIA
DEVELOPMENT OF ENVIRONMENTAL RELEASE OBJECTIVES
ESTABLISHMENT OF ATTESTATION REQUIREMENTS
MONITORING AND FOLLOW-UP
COMPLIANCE
yes
no
ENFORCEMENT PROCEDURE

Figure 1: Process to control the release of toxic substances into the aquatic environment from specific point sources.

RATIONALIZATION PROCEDURE FOR TOXIC SUBSTANCES


INDUSTRY
POLLUTANTS STUDIED FOR THE INDUSTRY CHOSEN
INFORMATION ON USES DOWNSTREAM FROM THE PLANT
ESTABLISHMENT OF QUALITY CRITERIA FOR THE POLLUTANTS STUDIED
CALCULATION OF DILUTION FACTORS (HYDRAULIC AND HYDROLOGIC CHARACTERISTICS)
PROBABLE EXCEEDANCE OF CRITERIA
NO
NO RECOMMENDATION FOR ANALYSIS
YES
SEARCH FOR A SITE OR A METHOD OF SAMPLING OR ANALYSIS
SUFFICIENT DETECTION LIMIT
PROBLEMS RESPECTING ANALYSIS ARISE IN LABORATORIES
YES
RECOMMENDATION OF PARAMETERS TO BE SAMPLED, TYPES OF SAMPLING AND SAMPLING SITES
SAMPLING RESULTS
MINIMUM TREATMENT TECHNOLOGIES
YES
CRITERIA MET
NO
ENVIRONMENTAL RELEASE OBJECTIVES FOR CRITICAL PARAMETERS
CORRECTIVE ACTION
DETERMINATION OF PROBLEMATIC AREAS
FOLLOW-UP OF IMPROVEMENT AND RECOVERY OF THE ENVIRONMENT

CRITERIA	USES	APPLICATION
Aquatic life acute toxicity	Aquatic life	end of pipe
Aquatic life Chronic toxicity	Aquatic life	Everywhere
Consumption of organisms	Fishing and consumption	Everywhere
Human health	Drinking water	Water intake
Terrestrial life	Terrestrial life	Everywhere
Aesthetics (taste and odour)		
Water	Swimming	Beach, kayak, etc.
Fish	Fishing	Everywhere

MIXING ZONE

ACUTE EFFECT

CHRONIC EFFECT

MAXIMUM 50 m OR ½ WIDTH

MAX 300 m

UPSTREAM

DIRECTION OF CURRENT

DOWNSTREAM

MAXIMUM DILUTION RATIO OF 1:100

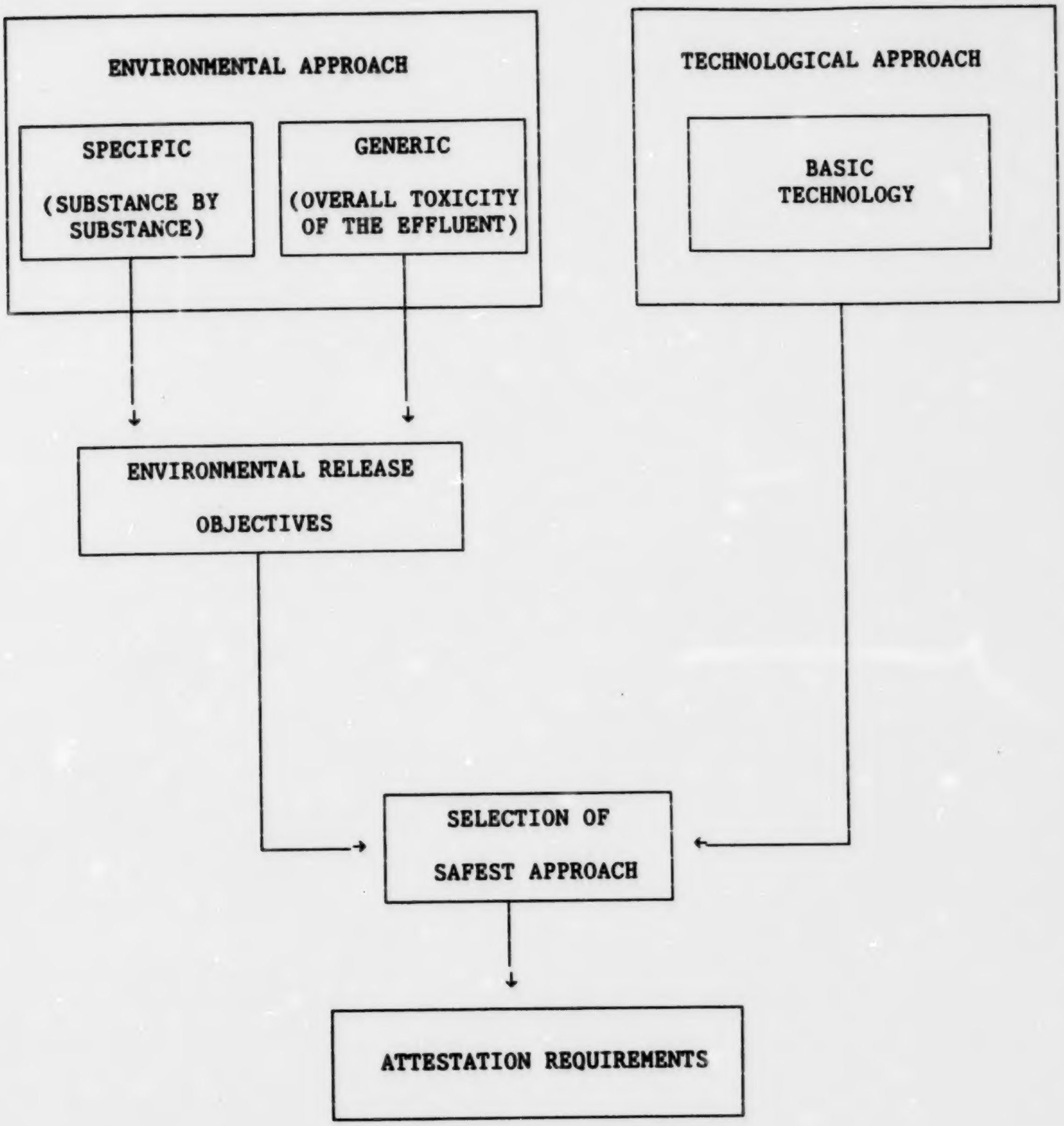


Figure 2: Definition of attestation requirements

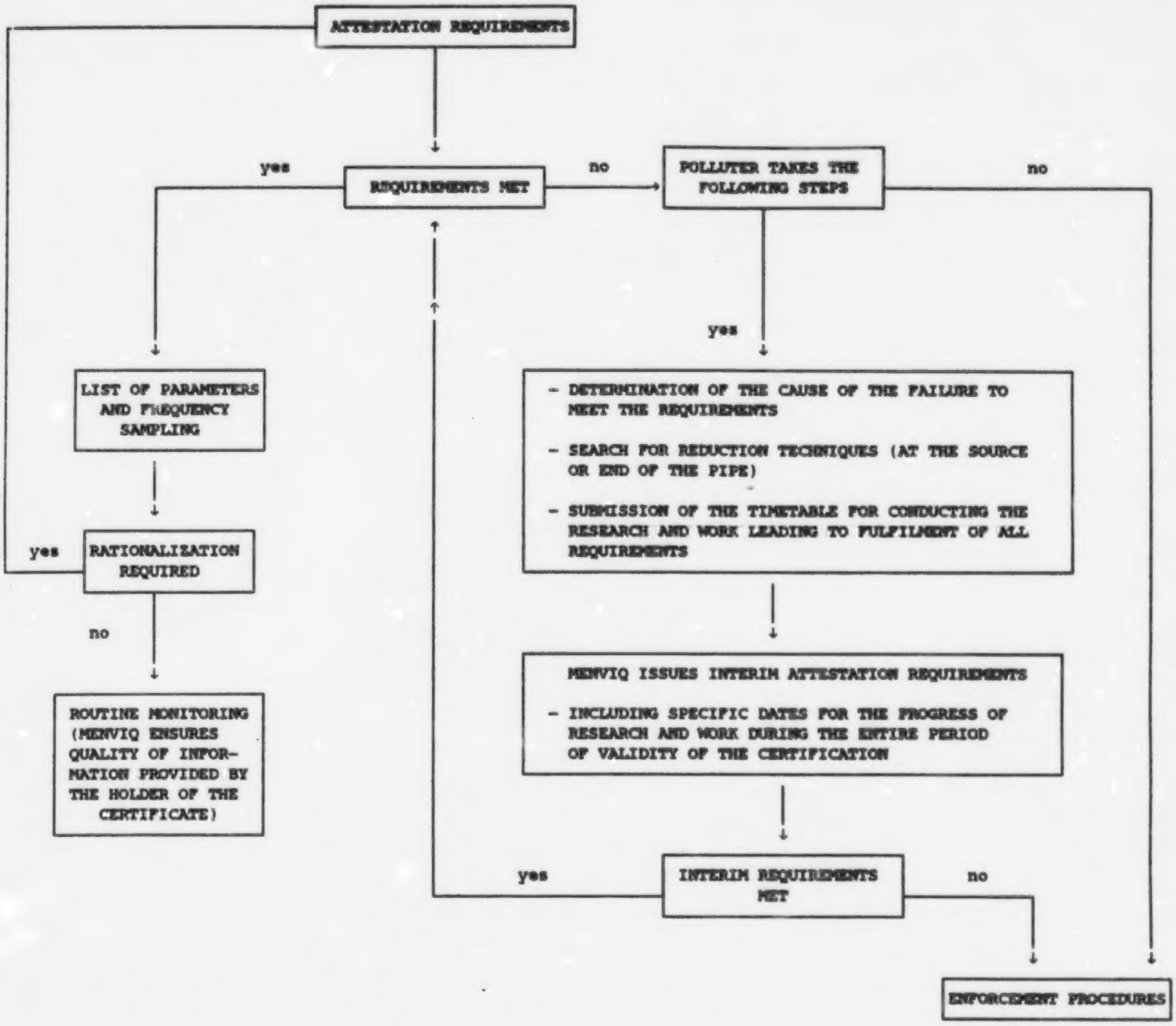

ATTESTATION REQUIREMENTS
yes
REQUIREMENTS MET
no
POLLUTER TAKES THE FOLLOWING STEPS
no
yes
LIST OF PARAMETERS AND FREQUENCY SAMPLING
- DETERMINATION OF THE CAUSE OF THE FAILURE TO MEET THE REQUIREMENTS
- SEARCH FOR REDUCTION TECHNIQUES (AT THE SOURCE OR END OF THE PIPE)
- SUBMISSION OF THE TIMETABLE FOR CONDUCTING THE RESEARCH AND WORK LEADING TO FULFILMENT OF ALL REQUIREMENTS
yes
RATIONALIZATION REQUIRED
no
MENVIQ ISSUES INTERIM ATTESTATION REQUIREMENTS
- INCLUDING SPECIFIC DATES FOR THE PROGRESS OF RESEARCH AND WORK DURING THE ENTIRE PERIOD OF VALIDITY OF THE CERTIFICATION
ROUTINE MONITORING (MENVIQ ENSURES QUALITY OF INFORMATION PROVIDED BY THE HOLDER OF THE CERTIFICATE)
yes
INTERIM REQUIREMENTS MET
no
ENFORCEMENT PROCEDURES

Figure 3: Process for the monitoring and follow-up of attestation requirements for the aquatic environment

1. MINIMUM ENVIRONMENTAL RISK

	Advantages	Disadvantages
Parameters/parameters	- acute toxicity	- does not take interactions into account
	- chronic toxicity	
	- bioaccumulable substances (partial)	- limited by knowledge about recognized substances
	- carcinogenic substances (partial)	
	- identification of possible causes	
Complete effluent	- takes interactions into account	- no identification of causes possible
		- bioaccumulation not covered
		- carcinogenicity not covered

Additions - organic scanning for bioaccumulable substances

- mutagenesis testing

2. Effective Direct Measures

- Preparation of lists of analyses
- Identification of sampling sites
- Identification of axes of analytical development of laboratories
- Establishment of better steps associated with follow-up in bodies of water

3. Clear Frame of Reference

- Establish clear complete objectives
- Identify the aspects of treatment design
- Identify problematic industrial processes and the focus of R&D on technology
- Permit the adequate allocation of financial resources to resolve actual environmental problems and find concrete solutions

4. General Aspects

- Legal aspects: tested procedure
- Industry must become involved and show scientific and technical evidence of their statements

Revised Water Quality
Objectives in Saskatchewan

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Regina, Saskatchewan
S4S 0B1

INTRODUCTION

Surface Water Quality Objectives (1988) published by Saskatchewan Environment and Public Safety, replace the 1983 edition. The water quality objectives were developed using available scientific literature and water quality criteria. The Canadian Water Quality Guidelines (CCREM 1987) were the major references along with several water quality criteria publications of the United States Environmental Protection Agency.

The Objectives are primarily oriented towards surface water quality. They are also applicable to ground waters which are utilized for specific purposes addressed in this report.

There are many instances where the natural water quality of a lake or river does not meet some of the objectives. In these cases, the objectives obviously will not apply. It should be noted, however, that where the natural existing quality is inferior to desirable objectives, it would be unwise to permit further deterioration by unlimited or uncontrolled introduction of pollutants. Naturally occurring circumstances are not taken into account in these "Objectives" and due consideration must be given where applicable (e.g. spring runoff effect on colour, odour; ice and snow cover effect on dissolved oxygen and influences on major ion levels; rainfall influences on bacteria levels in surface waters).

In future years, the Department will be revising these objectives as improved knowledge of various constituents/conditions becomes available. Also additional use-specific objectives and basin specific objectives may be developed for different basins characterized by different water quality characteristics.

The 1988 Objectives contain the following sections:

1. General Policies

2. Uses of Water Quality Objectives

3. Objectives for Effluent Discharges

    3.1 General Objectives

    3.2 Guidelines for Effluent Mixing Zones

4. **Surface Water Quality Objectives**
    - 4.1 **General Surface Water Quality Objectives**
    - 4.2 **Specific Surface Water Quality Objectives**
        - 4.2.1 Aquatic Life and Wildlife
        - 4.2.2 Recreational Uses
            - 4.2.2.1 Contact Recreation
            - 4.2.2.2 Non-Contact Recreation
        - 4.2.3 Agricultural Uses
            - 4.2.3.1 Crop Irrigation
            - 4.2.3.2 Livestock Watering
        - 4.2.4 Potable Water Supply

5. Algae amd Aquatic Nuisances

6. Information Sources

References

Explanatory Notes Regarding Water Quality Terms/Parameters.

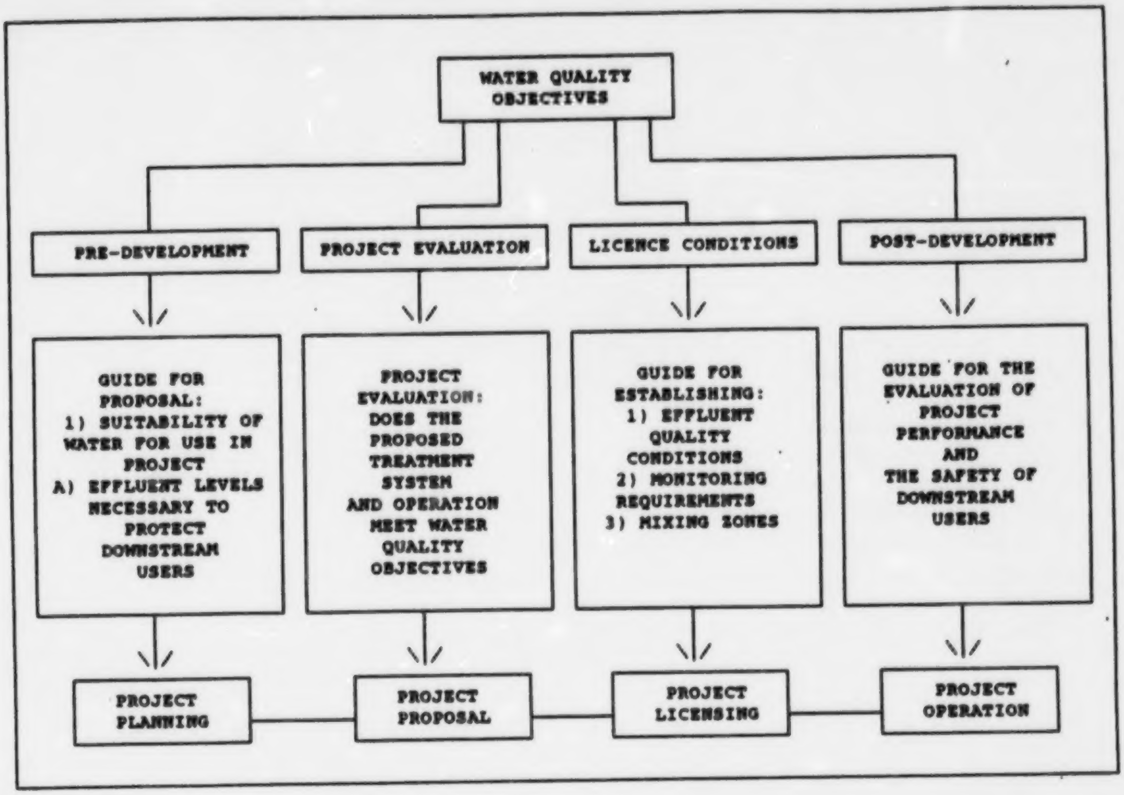
The booklet "Surface Water Quality Objectives" can be obtained from:

Water Quality Branch
Saskatchewan Environment and Public Safety
3085 Albert Street
Regina, Saskatchewan
S4S 0B1
[Phone (306)787-6238]

Municipal drinking water quality objectives are contained in a separate booklet published by the Department.

REFERENCE

CCREM. 1987. Canadian Water Quality Guidelines. Prepared by the Task Force on Water Quality Guidelines of the Canadian Council of Resource and Environment Ministers. March 1987.


WATER QUALITY OBJECTIVES
PRE-DEVELOPMENT
PROJECT EVALUATION
LICENCE CONDITIONS
POST-DEVELOPMENT
GUIDE FOR PROPOSAL: 1) SUITABILITY OF WATER FOR USE IN PROJECT A) EFFLUENT LEVELS NECESSARY TO PROTECT DOWNSTREAM USERS
PROJECT EVALUATION: DOES THE PROPOSED TREATMENT SYSTEM AND OPERATION MEET WATER QUALITY OBJECTIVES
GUIDE FOR ESTABLISHING: 1) EFFLUENT QUALITY CONDITIONS 2) MONITORING REQUIREMENTS 3) MIXING ZONES
GUIDE FOR THE EVALUATION OF PROJECT PERFORMANCE AND THE SAFETY OF DOWNSTREAM USERS
PROJECT PLANNING
PROJECT PROPOSAL
PROJECT LICENSING
PROJECT OPERATION

PRESENT AND FUTURE WATER QUALITY OBJECTIVES' ACTIVITIES IN MANITOBA

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INTRODUCTION

The Manitoba Department of Environment developed a proposal detailing a system of surface water quality objectives and watershed classifications for the Province of Manitoba in 1976. The original proposal was modified slightly following widespread public review (Clean Environment Commission 1979). This system was then used to classify three major watersheds in Manitoba according to water use, namely the Souris, Red and Grass-Burntwood rivers' basins (Clean Environment Commission 1980, 1981, 1982). A number of major technical revisions were proposed for the program in 1983 (Williamson 1983a, 1983b). These proposed revisions were the subject of widespread scientific, technical and public review. The public review culminated with a two-day public hearing, held in Winnipeg during November, 1984. Several additional revisions were made subsequent to these public hearings and the final document was released on July 31, 1988 (Williamson 1988a). Accompanying this final Manitoba Surface Water Quality Objectives report was a rationale document describing the reasons for the latter series of revisions, including an explanation of the most controversial parts of the program, and containing the Clean Environment Commission's report on the findings of their public hearings held in 1984 (Williamson 1988b). Throughout this entire period, the Manitoba Surface Water Quality Objectives were used extensively for water quality management activities. Principal amongst these was their use by developers as a planning tool and by Manitoba Environment to assist in developing effluent discharge limitations in legally-binding licences. The Environment Act, enacted in Manitoba in March, 1988, provides legislative support for the Department's present role in the development and implementation of water quality objectives. Government-wide support for their development and implementation is provided in the emerging provincial water strategy (Anonymous 1989).

Other Canadian and international jurisdictions define water quality objectives in various terms. Terms such as criteria, guidelines, objectives, standards, site-adapted guidelines, site-specific guidelines, site-specific objectives, plus others are in common use. Of these various terms, only water quality standards are legally enforceable. Water quality standards are required by U.S. federal law for all state jurisdictions within the United States. All terms generally represent a sequential refinement of the initial scientific toxicological information to fit the specific circumstances at a single site. Within Manitoba, the following definition applies:

> The Manitoba Surface Water Quality Objectives define minimum levels of quality necessary for the protection of the important water uses in Manitoba. The objectives, when not exceeded, will protect an organism, a community of organisms, or other designated multiple-purpose water uses.

Thus, the Manitoba Surface Water Quality Objectives are analogous to the Canadian Water Quality Guidelines [Canadian Council of Resource and Environment Ministers (CCREM) 1987], with the exception that they have been site-adapted for general use in Manitoba.

A summary of this program and its use in Manitoba is described in the following sections. The general process, from the initial scientific research through to the actual use of water quality objectives in Manitoba water management activities, is illustrated in Figure 1. Linkages with agencies outside Manitoba Environment are indicated at the bottom of the figure.

SCIENTIFIC RESEARCH

Scientific research on cause-effect relationships between concentrations of a toxic material and aquatic organisms, environmental transport phenomena, and fate within the aquatic ecosystem forms the basis for all credible water quality objectives. Primary research is conducted by scientists throughout the world, with information being available through scientific journals. In most cases, problems encountered in other jurisdictions are similar either to those existing at present or those that could potentially exist in the future within Manitoba. Thus, with few exceptions, the world scientific toxicological data base is generally suitable for meeting the information needs of Manitoba. Therefore, Manitoba Environment does not conduct primary scientific research in this area, but relies upon information produced by other research agencies.

DEVELOPMENT OF WATER QUALITY OBJECTIVES

Manitoba Environment does not directly develop objectives for the various materials of concern from the primary scientific literature. Rather, criteria, guidelines, objectives or standards from other jurisdictions are thoroughly evaluated for their applicability to Manitoba, then adopted if appropriate. In a number of cases, the information has been modified to better suit the unique conditions within Manitoba. This method is efficient in terms of utilizing limited financial and human resources.

Water quality objectives (or criteria, guidelines, etc.) have been developed for numerous variables from basic research using systematic methods by a number of jurisdictions and agencies [U.S. EPA 1985, CCREM 1987, CCME 1989, Ontario Ministry of Environment 1989, European Inland Fisheries Advisory Commission (Alabaster and Lloyd, 1982), Health and Welfare Canada 1987, plus others]. Such systematic methods differ somewhat between agencies, depending upon philosophical and scientific approaches related to the degree of protection desired within each jurisdiction and the desired level of confidence required in the data base. Manitoba Environment (Williamson 1988a) has selected the methods used by the U.S. EPA (1985) for the protection of aquatic life and wildlife for those variables with a large existing toxicological data base. Objectives for other materials are selected from other jurisdictions when the U.S. EPA (1985) minimum data base requirements cannot be satisfied. Thus, Manitoba Environment has adopted the philosophy of preventing the occurrence of unacceptable levels of impairment as opposed to preventing the occurrence of all observed responses (Williamson 1988b).

Manitoba Surface Water Quality Objectives are in two principal forms, namely general or narrative surface water quality objectives and specific or numerical surface water quality objectives. Additionally, guidance is provided through a number of policies that assist in the implementation and interpretation of these objectives in water quality management activities. Each of these three areas are described in the following sections.

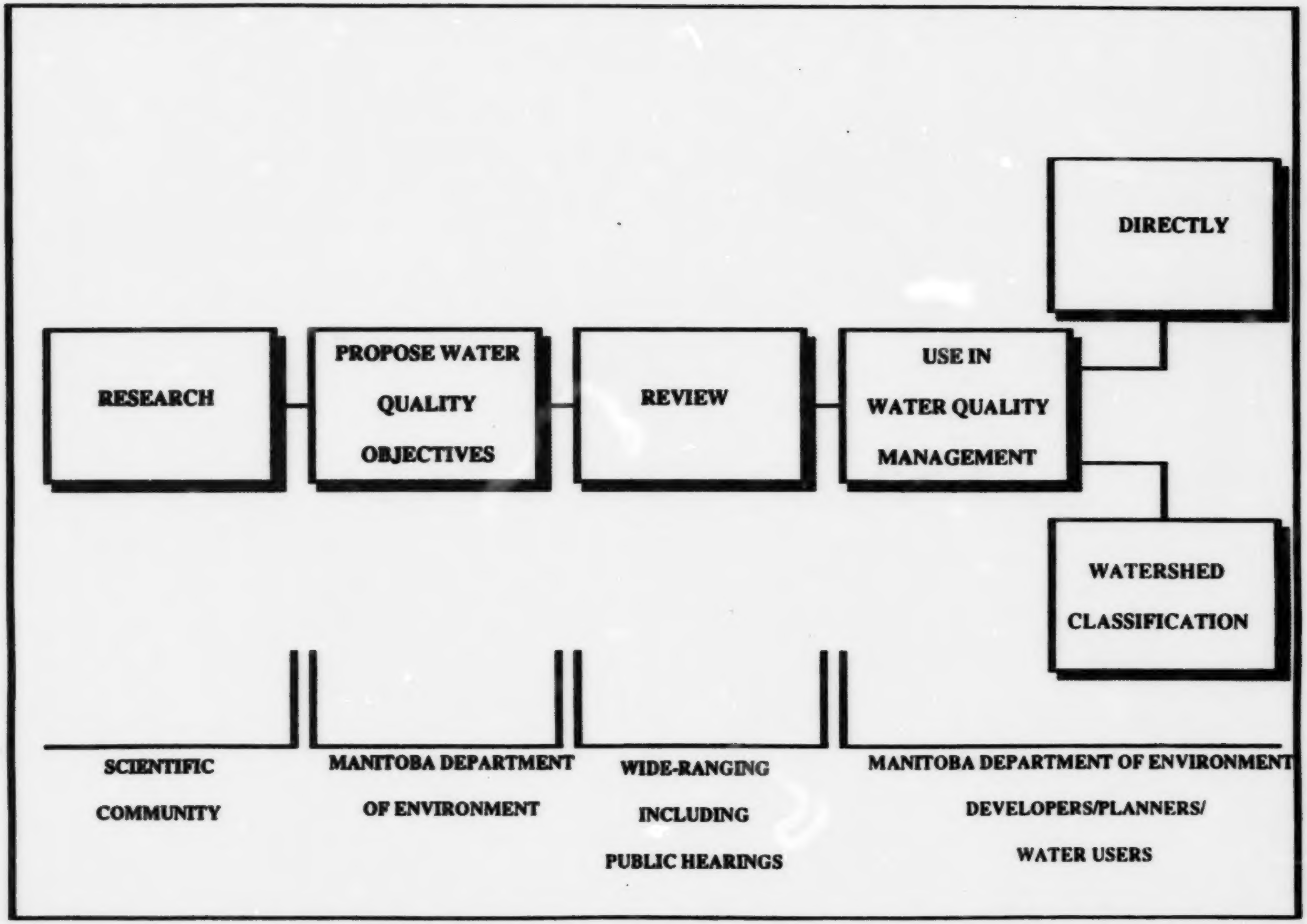
RESEARCH
PROPOSE WATER QUALITY OBJECTIVES
REVIEW
USE IN WATER QUALITY MANAGEMENT
DIRECTLY
WATERSHED CLASSIFICATION
SCIENTIFIC COMMUNITY
MANITOBA DEPARTMENT OF ENVIRONMENT
WIDE-RANGING INCLUDING PUBLIC HEARINGS
MANITOBA DEPARTMENT OF ENVIRONMENT
DEVELOPERS/PLANNERS/ WATER USERS

Figure 1: General process used in the development and use of Manitoba Surface Water Quality Objectives. Linkages are shown in the lower portion of the figure.

(1). NARRATIVE SURFACE WATER QUALITY OBJECTIVES

Narrative surface water quality objectives are general statements that attempt to describe conditions within Manitoba's aquatic systems that should be avoided. While such narrative statements are of less practical value than are specific numerical surface water quality objectives, the lack of scientific information precludes the development of more specific objectives at the present time. These narrative surface water quality objectives provide statements of intent, that can be used to protect water quality. Such narrative statements have been developed for colour, odour, taste, turbidity, floating materials, oil and grease, deposits on bottom sediments or shorelines, nutrients, toxic substances, litter, and flow. For example, the narrative surface water quality objective for oil and grease is as follows:

> "Free from oil and grease residues which causes a visible film or sheen upon the waters or any discolouration of the surface of adjoining shorelines or causes a sludge or emulsion to be deposited beneath the surface of the water or upon the adjoining shorelines."

(2). NUMERICAL SURFACE WATER QUALITY OBJECTIVES

Numerical surface water quality objectives have been adopted from other jurisdictions and in some cases have been site-adapted for use in Manitoba. Such objectives are included for over 80 variables. The numerical surface water quality objectives are arranged according to the important water uses within Manitoba. These uses are similar to those described by other jurisdictions (cf. CCREM 1987, Saskatchewan Environment and Public Safety 1988). Manitoba's numerical surface water quality objectives are arranged to protect the following uses:

Class 1: Domestic Consumption: Numerical surface water quality objectives will protect the use of raw water supplies for domestic consumption, culinary, food processing or other household purposes. Inherent in these objectives is the necessity to disinfect all raw surface water supplies prior to use as a minimum level of treatment.

Class 2: Aquatic Life and Wildlife: The numerical surface water quality objectives within this class will ensure the protection of waters that are inhabited by aquatic life such as fish, amphibians, reptiles, and other forms of life including aquatic insects and algae. By ensuring protection of the aquatic communities, protection is indirectly offered to those forms of wildlife that rely upon surface waters for habitat and for food supplies. These include ducks, geese, fur-bearing mammals such as the muskrat and birds of prey such as the eagle and osprey. Protection is also provided to those animals that use these waters for drinking purposes.

This class is divided into two categories in order to provide protection to the two general types of aquatic life communities within Manitoba. The first category will provide protection to all types of aquatic life in Manitoba including wildlife that rely upon surface waters for habitat or for a source of food supplies. This includes both "cold-water" and "cool-water" types of aquatic life communities. The second category will only provide protection to "cool-water" types of aquatic life communities, as well as wildlife.

Class 3: Industrial Consumption: Objectives within this class will protect the quality of raw water sources used for industrial purposes. However, because the various general types of industries have unique water quality requirements, numerical objectives have not yet been adopted for this class. Rather, the unique requirements of industries are considered on a site-specific basis during assessments and licencing of upstream dischargers. Generally, guidelines for industrial water quality, developed by the CCREM (1987) are used for this purpose.

Class 4: Agricultural Consumption: This class is divided into four categories, three of which are intended to protect the quality of surface water used for three general types of irrigation. These include the irrigation of greenhouse plants, plus two types of field crop irrigation. The fourth category is intended to protect the quality of water used for livestock watering.

Class 5: Recreation: This class is divided into two categories in order to provide protection for the quality of water used for two general types of aquatic recreation. The objectives within the PRIMARY RECREATION category are intended to protect surface water used for activities in which the entire body may become completely immersed. The objectives within the SECONDARY RECREATION category will protect the quality of water used for other water activities, such as boating, where there is considerably less probability that the entire body would become completely immersed.

Class 6: Other Uses: This class is set aside for those surface water uses that require protection but are not well defined within the preceding classes or categories. Additionally, certain waters, for example, may be set aside for incorporation into industrial treatment processes. Such waters, therefore, may not require protection of water quality.

A brief example of the numerical surface water quality objectives is shown in Table 1.

Table 1: Numerical surface water quality objectives for five common variables for the protection of water uses in Manitoba. Additional categories are listed in the Manitoba Surface Water Quality Objectives but are not illustrated in this example.

VARIABLE	DOMESTIC CONSUMPTION	AQUATIC LIFE AND WILDLIFE	AQUATIC LIFE AND WILDLIFE	GREENHOUSE IRRIGATION	FIELD CROP IRRIGATION	LIVESTOCK WATERING	PRIMARY RECREATION	SECONDARY RECREATION
Fecal Coliform	10/100 mL	No Obj.	No Obj.	1000/100 mL	1000/100 mL	No Obj.	200/100 mL	1000/100 mL
Sodium	400.0 mg/L	No Obj.	No Obj.	4.0 SAR	6.0 SAR	No Obj.	No Obj.	No Obj.
Boron	5.0 mg/L	No Obj.	No Obj.	0.5 mg/L	1.0 mg/L	5.0 mg/L	No Obj.	No Obj.
Dissolved Oxygen	No Obj.	60% Sat.	47% Sat.	No Obj.	No Obj.	No Obj.	Aerobic	Aerobic
Zinc	5.0 mg/L	0.047 mg/L	0.047 mg/L	2.0 mg/L	10.0 mg/L	50.0 mg/L	No Obj.	No Obj.

(3a). IMPLEMENTATION POLICIES-LIMITATIONS AND MODIFICATIONS, MIXING ZONES, AND LOW FLOW CONDITIONS

Information is provided in Williamson (1988a) to assist in the application of the Manitoba Surface Water Quality Objectives at specific sites. This information is in the form of a number of policies or guidelines concerning limitations of the objectives, how and under what general circumstances modifications can be made to the objectives, how the objectives should apply to point-source discharges within the area of initial effluent mixing, and how the objectives apply to extremely low river and stream flow.

With regard to mixing zones, it is recognized that it is not reasonable in many circumstances to expect the numerical surface water quality objectives to be met at the distal end of the discharge pipe. Therefore, a zone of initial dilution is necessary in which some numerical objectives may not be met. Therefore, guidelines were developed to limit the nature and extent of adverse conditions within this zone. The following are several examples:

> "the mixing zone should be designed to allow an adequate zone of passage for the movement or drift of all stages of aquatic life,
>
> (i). for those materials that elicit an avoidance response from aquatic life, the mixing zone should contain not more than 25% of the cross-sectional area or volume of flow at any transect in the receiving water. Should a proportion of the stream width greater than 25% be selected for these materials, the mixing zone could act similar to a physical barrier and could effectively preclude the passage of aquatic life,
>
> (ii). the mixing zone should not be acutely lethal to aquatic life passing through the mixing zone. Thus, acute lethality within the mixing zone is a function of the concentration of a toxic material and the duration of exposure,
>
> (iii). mixing zones should not interfere with the migratory routes essential to the reproduction, growth or survival of aquatic species,
>
> (iv). mixing zones should not cause an irreversible organism response, or increase the vulnerability to predation,
>
> (v). when two or more mixing zones are in close proximity, they should be so defined that a continuous passageway for aquatic life is available,
>
> (vi). mixing zones should not intersect the mouths' of rivers,"

It is generally recognized that it is not practicable for dischargers to comply with the numerical surface water quality objectives under all possible low-flow situations on specific rivers or streams. Thus, a policy was developed which requires that the objectives must be met at all river and stream flows above the average minimum seven day flow which occurs once in ten years (Q_{7-10}). Exceptions include cases where the flow is less than this volume but important water uses are still maintained by the effect of ponding. Intermittent streams which flow for a short period most years, have a Q_{7-10} of 0.0 m^3/s. Since the Q_{7-10} is of little practical value in these cases, a policy was developed that requires the numerical surface water quality objectives to be met on intermittent streams at all flows above 0.003 m^3/s.

(3b). IMPLEMENTATION POLICIES-LEVELS OF PROTECTION

The Manitoba Surface Water Quality Objectives offer options for four different levels of protection to specific aquatic systems. These are as follows:

NO PROTECTION: Circumstances at limited sites may determine that a specific body of water should not be afforded protection, or an individual water use within that body of water should not be afforded protection. These are legitimate decisions that can be made by water quality managers, based upon scientific, social or economic constraints. For example, a small lake or marsh may become incorporated into the wastewater treatment system at an industrial processing site. Hence, such a body of water would not be afforded protection.

ROUTINE: The routine level of protection within Manitoba is a combination of firstly, the application of Best Available Technology Economically Achievable (BATEA), more commonly known in Canada as Best Practicable Technology (BPT), and secondly, the application of the Manitoba Surface Water Quality Objectives in cases where water uses are not adequately protected by the sole use of BATEA or BPT. This ensures that all pollutants are reduced or eliminated with the use of standard treatment technologies commonly available to each unique industry. It also recognizes that in other cases, the sole use of common, technology-based treatment systems may not provide protection to a specific body of water (e.g. large volume of discharge to a small stream, a number of industries discharging to a single body of water, etc.). In these cases, surface water quality objectives are used to develop effluent limitations that will provide the required protection. The effluent limitations are developed such that the surface water quality objectives will not be exceeded.

HIGH QUALITY DESIGNATION: A major limitation associated with the above mentioned routine level of protection is that all bodies of water are considered equal in value (intrinsic, economic and social) and that some magnitude of change is acceptable. Ultimately, the change could be greater than that expected, thus resulting in an adverse impact. The routine method also places little emphasis upon maintaining the existing quality of a body of water. Therefore, the HIGH QUALITY DESIGNATION can be used to place greater restrictions upon development within certain pristine or near-pristine areas of Manitoba.

The HIGH QUALITY DESIGNATION recognizes that some development has already occurred within the region in question or that future development may be probable. Despite this, it is desirable to place greater environmental restrictions on new or expanded developments. The water bodies within these areas must meet certain requirements in order to be considered for this designation. These include the following:

(a). waters that flow through or that are bounded by Provincial or National Parks,
(b). waters within relatively undisturbed watersheds,
(c). waters possessing outstanding quality characteristics,
(d). waters that support a diverse or unique flora and fauna which are sensitive to man-induced water quality alterations,
(e). waters designated as Canadian Heritage Rivers.

The greater restrictions placed upon developments within HIGH QUALITY areas include firstly, an additional degree of justification of the proposed new, additional or increased discharges in terms of the social and economic benefits to the local, regional or Manitoban economies. Secondly, should the proposed development be of significant benefit and alternatives not be available in other nearby regions, then a combination of Best Available Technology, land disposal, re-use and discharge technologies will be used. Thus, developers can anticipate much higher costs associated with environmental control within areas designated as HIGH QUALITY.

This designation will ensure that the risk of altering water quality will be maintained at zero or as close to zero as possible, but it also recognizes that some development has already occurred, and that in future other developments might similarly be considered. The designation does not prohibit all such developments, but requires that they satisfy certain stringent criteria.

Water bodies are designated as HIGH QUALITY through the WATERSHED CLASSIFICATION process discussed in a subsequent section. At present, over 10,000 km^2 of Manitoba have been designated as HIGH QUALITY.

EXCEPTIONAL VALUE DESIGNATION: This designation provides the fourth and final level of water quality protection within Manitoba. Water courses designated as such will not receive "any alterations that result in measurable, calculable or perceived water quality degradation or degradation of other values deemed exceptional" (Williamson 1988a). Waters considered for this designation are as follows:

(a). Ecological Reserves,
(b). wild and scenic rivers or lakes,
(c). waters or watersheds providing habitat for rare or endangered flora and fauna,
(d). waters considered sensitive such that irreversible harm will result following human impact,
(e). waters whose exceptional quality and value as a future resource precludes the assignment of present uses,
(f). waters designated as Canadian Heritage Rivers.

Water bodies designated as EXCEPTIONAL VALUE, will be essentially removed from the potential for development, thus eliminating the risk of water quality change due to man's local influence. The EXCEPTIONAL VALUE designation will be applied through the WATERSHED CLASSIFICATION process. At present, no such water bodies have been designated as EXCEPTIONAL VALUE, although a number of potential candidates exist in remote northern and eastern areas of Manitoba.

SCIENTIFIC, TECHNICAL AND PUBLIC REVIEW

A large scientific, technical and public review component is involved in the Manitoba Surface Water Quality Objectives. Initially, recommendations concerning the objectives are made by staff within Manitoba Environment, then are circulated for review and comment to a standing committee comprised of representatives from numerous other Manitoba government departments. At present, the Interdepartmental Committee on Manitoba Surface Water Quality Objectives consists of 13 members from other Manitoba government agencies. This participation ensures that the objectives reflect a wide perspective within the Manitoba provincial government. The objectives are then circulated to a large number of outside agencies representing research scientists, environmental groups, specific industries and municipalities, industrial and municipal associations, and other Canadian jurisdictions. Additionally, copies are forwarded to the Clean Environment Commission, to ensure that widespread public involvement is obtained. The Clean Environment Commission, appointed by Cabinet and chaired by a full-time civil servant, solicits input from the public and provides independent advice to the Minister, as provided for in the Manitoba Environment Act. If the Clean Environment Commission or Manitoba Environment deems it necessary, public hearings are held. The Clean Environment Commission held public hearings on the original surface water quality objectives proposal during the late 1970's and held public hearings on a major series of revisions in 1984.

USE OF MANITOBA SURFACE WATER QUALITY OBJECTIVES IN WATER QUALITY MANAGEMENT ACTIVITIES

The Manitoba Surface Water Quality Objectives are used for the following purposes:

(a). **AS A PLANNING TOOL:** Using the Manitoba Surface Water Quality Objectives, developers can make informed decisions with regard to the type and cost of environmental control required at the preferred site in comparison to others. This information can be used very early in project planning, just as information on transportation costs, availability of land and its costs, work-force costs, and other important factors must be considered. Potential developers can therefore quite accurately estimate the environmental control costs associated with any proposed location.

(b). **TO DEVELOP EFFLUENT DISCHARGE LIMITS:** Virtually all projects that have the potential to impact the environment must be licenced under the Manitoba Environment Act. For example, in cases where the ROUTINE level of protection is being provided, as described above, Manitoba Department of Environment use the surface water quality objectives to limit the discharge of materials not sufficiently controlled by the use of Best Available Technology Economically Available-based treatment systems. The general factors that must be considered are shown in Figure 2. A thorough

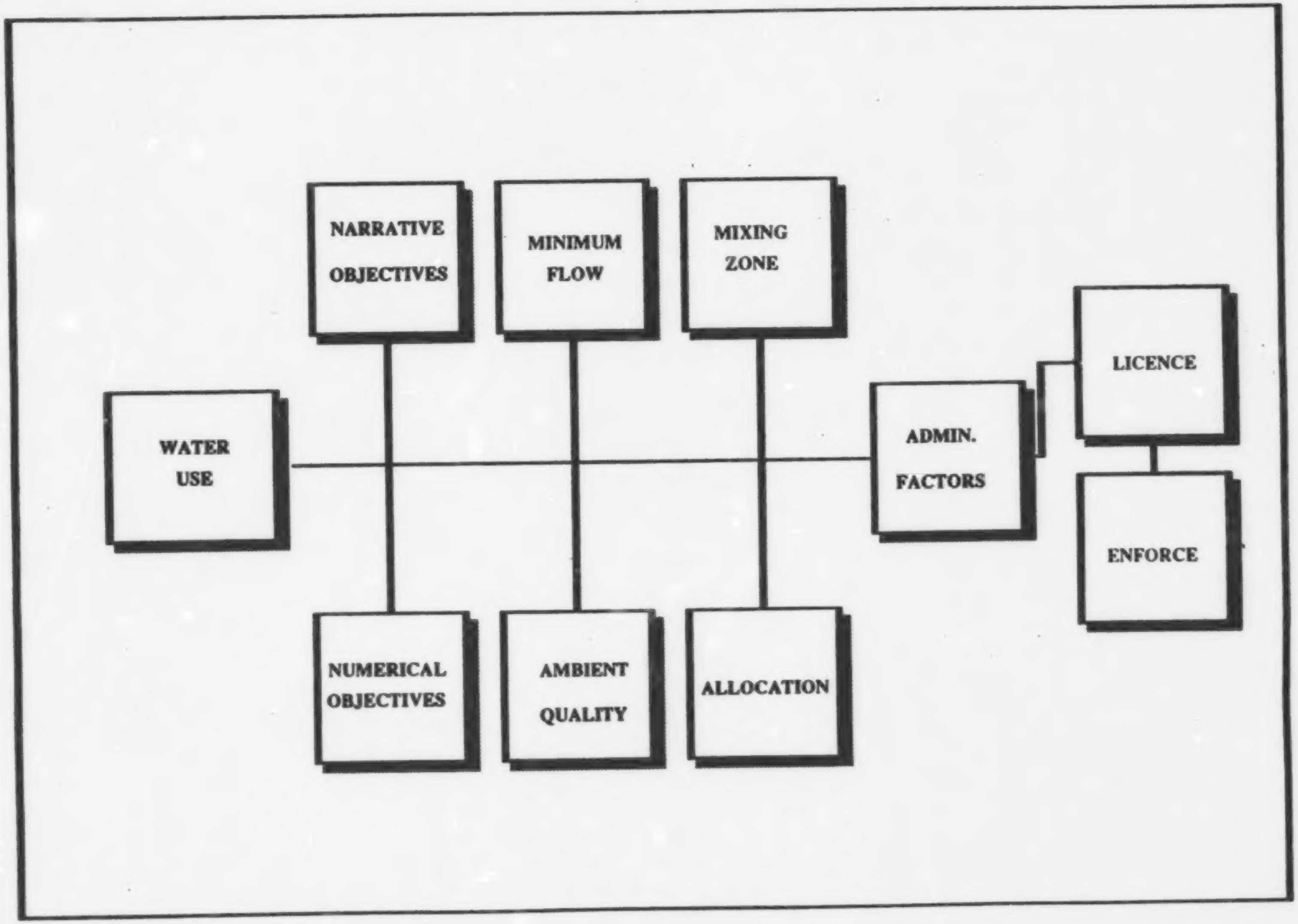

Figure 2: Process diagram showing the information required to derive effluent discharge limitations from the Manitoba Surface Water Quality Objectives.

assessment is conducted of the proposed facility, including the identification of all potential pollutants and their expected concentration or discharge load. The assessment will then address the following:

(i). identification of the downstream water uses,
(ii). the narrative and numerical surface water quality objectives for the identified pollutant necessary in order to protect the downstream water uses,
(iii). the minimum stream flow (Q_{7-10}),
(iv). delineation of a mixing zone for the pollutant in question. Should the material cause an avoidance response, an unobstructed pollutant-free passage within the stream will be maintained.
(v). allocation of the available dilution or assimilative capacity for the proposal. A proportion of the assimilative capacity may be left un-allocated for either future expansion or for future downstream developments.

Maximum allowable loadings or concentrations are then computed using commonly accepted models. Most frequently, such models are simple algorithms based upon conservation of mass, without consideration of transformation or partitioning within the receiving body of water. In some cases, the result may be modified slightly in order to incorporate the information into a legally-enforceable licence.

This process is further described in the brief example shown in Figure 3. Springhill Farms Ltd., a large hog abbatoir, is located near the Town of Neepawa, in the south-central region of Manitoba [for general reference, this area is located near the southern-most region of the Dauphin River Watershed (Watershed #5) on Figure 5]. Liquid effluent discharges are directed towards the Whitemud River, a relatively small stream that eventually empties into Lake Manitoba. The stream is used as a source of water for domestic consumption, irrigation, livestock watering, recreation and habitat for native cool water and stocked cold water species of fish. Water flow in this stream is regulated by a control structure at the outlet of Lake Irwin. The Town of Neepawa also discharges treated municipal sewage twice annually, from a multi-celled lagoon, to the Whitemud River immediately upstream from Springhill Farms Ltd.

A preliminary environmental assessment identified un-ionized ammonia discharges from the development as a potential concern. During a detailed assessment, monthly limitations were calculated for un-ionized ammonia. The information that formed the basis of the limitations for the month of July is as follows (please note that while this is an actual example, the information considered during the assessment is much too lengthy to be adequately documented here):

Minimum stream discharge	= 17122 m^3/day (secured by the proponent through a Water Rights Licence)
Average July pH	= 8.04
Average July water temperature	= 17.9°C
Average upstream un-ionized ammonia	= 0.0006 mg/L

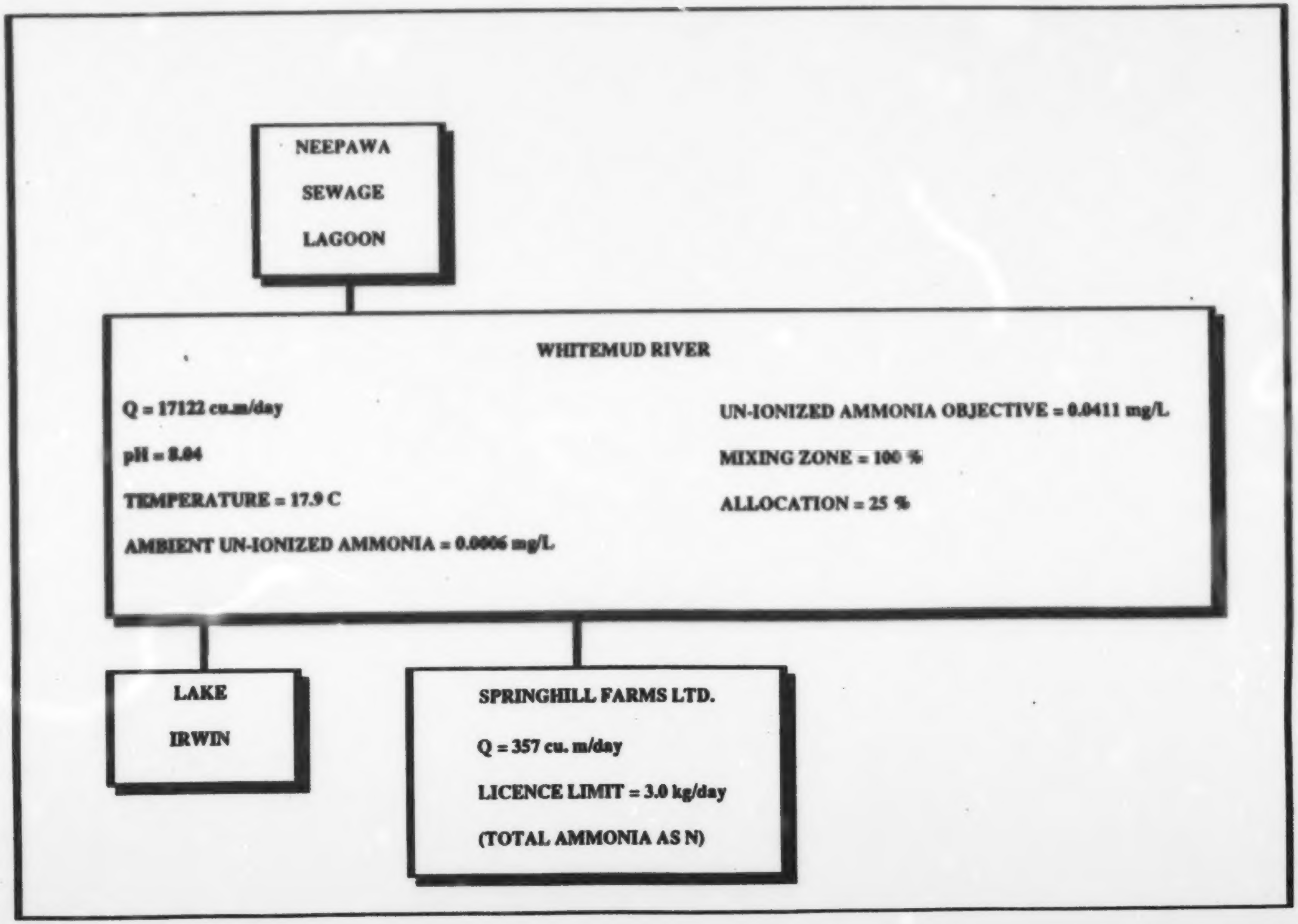

Figure 3: Actual example illustrating the use of the Manitoba Surface Water Quality Objectives to derive effluent discharge limitations for un-ionized ammonia at Springhill Farms Ltd. for the month of July.

Predicted volume of effluent discharge = 357 m^3/day

Manitoba Surface Water Quality Objective for un-ionized ammonia at 17.9°C and 8.04 pH units = 0.0411 mg/L

Mixing zone = 100% of stream width or volume of flow

Allocation of assimilative capacity = 25% of volume of flow

Limitation in Licence 1103VC = 3.0 kg/day, total ammonia as N (calculated using conservation of mass algorithm after appropriate conversions between un-ionized ammonia and total ammonia using accepted equilibrium equations)

(c). **TO DEVELOP LAND MANAGEMENT PRACTICES:** While more technically difficult than the use of the objectives to develop effluent discharge limitations, it is possible to develop land management plans aimed at reducing diffuse sources of pollutants to water bodies.

(d). **TO INTERPRET WATER QUALITY DATA:** Ambient water quality data can be compared directly to the objectives to identify exceedences or long-term trends that may lead to exceedences in the future. Management intervention can then be undertaken.

(e). **TO ASSESS CONTROL EFFECTIVENESS:** The surface water quality objectives can be used to determine whether or not effluent limitations are effective in ensuring that downstream objectives are not exceeded.

(f). **TO DETERMINE WHETHER SPECIFIC WATERS ARE SUITABLE FOR CERTAIN USES:** The surface water quality objectives, in combination with ambient monitoring data, can be used to initially determine whether or not specific bodies of water are suitable for certain proposed uses or activities.

WATERSHED CLASSIFICATION

The Manitoba Surface Water Quality Objectives can be used directly as discussed in the previous section, or they can be used to develop long-term water quality management plans, through WATERSHED CLASSIFICATION. WATERSHED CLASSIFICATION is an administrative and planning process whereby surface water quality objectives are applied to individual streams, lakes, or entire watersheds in order to protect multiple water uses. The surface water quality objectives may be modified, where appropriate, to better reflect the unique circumstances within the area under consideration. Modifications can be undertaken to account for the lower or greater sensitivity of resident aquatic species, to consider the altered availability or toxicity of a pollutant due to chemical or physical properties of the receiving water, or other reasons. Reasonable scientific evidence, professional judgement, or other evidence must be used to ensure that surface water uses at specific sites will not be unacceptably impaired following such modifications. WATERSHED CLASSIFICATION is a complex process and is outlined in Figure 4.

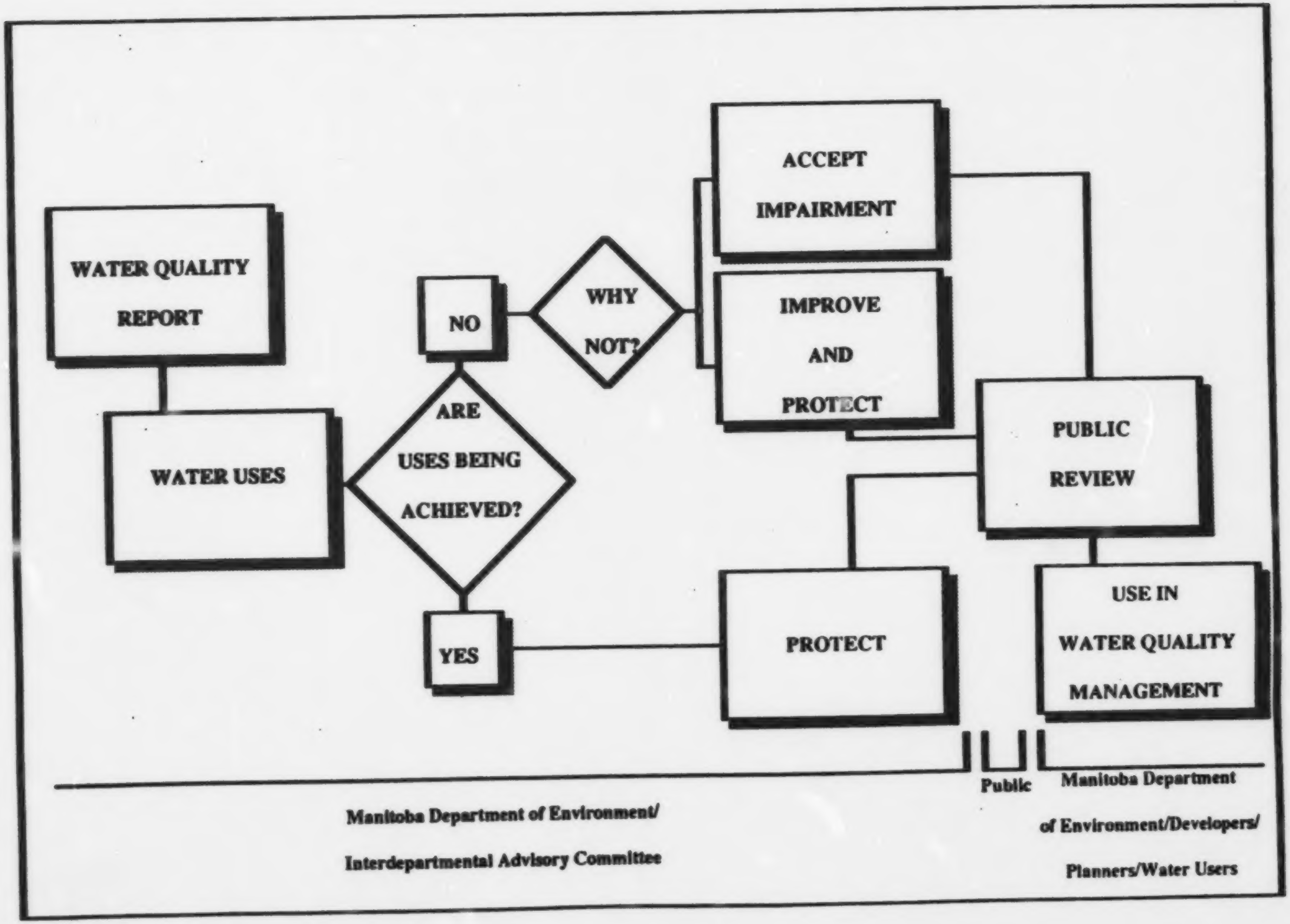

Figure 4: Diagram illustrating the WATERSHED CLASSIFICATION process within Manitoba.

A water quality report is prepared that consists of an assessment of existing conditions, identification of existing and potential future water uses, and determination of whether or not water quality may be a limiting factor in achieving the present or future uses. If water quality is not a limiting factor, then water quality objectives are recommended to protect the identified uses. An appropriate level of protection (e.g. ROUTINE, HIGH QUALITY, or EXCEPTIONAL VALUE) is also recommended during this stage of the process. However, if the existing water quality is presently impaired, thus impacting either a present or future water use, further evaluation is necessary. This additional evaluation is guided by the following general questions:

(1). Which water uses are being impaired?
(2). What are the water quality variables causing the impaired use?
(3). To what extent does human activities contribute to the impairment?
(4). What level of control is required to ameliorate the water quality exceedences?
(5). Do control technologies actually exist in order to achieve the level of reclamation necessary?
(6). Does the cost of achieving the water quality improvement bear a reasonable relationship to the benefits associated with attaining the water use?

Depending upon the result of this evaluation, surface water quality objectives could be recommended for the site under consideration such that the impaired water quality would be accepted. Alternately, objectives could be recommended that would provide the basis for a plan that would improve the impaired water quality to the level necessary to protect the presently impacted water use.

Following the preparation of the WATERSHED CLASSIFICATION proposal, it is subject to wide-ranging technical, scientific and public review, including public hearings conducted by the Clean Environment Commission. This review is similar to that discussed in the preceding section entitled SCIENTIFIC, TECHNICAL AND PUBLIC REVIEW. Following endorsement of the finalized proposal by government, the surface water quality objectives that are refined and established through the WATERSHED CLASSIFICATION process, are used as the basis for managing water quality, as identified in the preceding section entitled USE OF MANITOBA SURFACE WATER QUALITY OBJECTIVES IN WATER QUALITY MANAGEMENT ACTIVITIES.

WATERSHED CLASSIFICATION offers a number of advantages in comparison to direct use of the surface water quality objectives. Principal amongst these is the ability to develop a rational, long-term, water quality plan for a region, in a public process that encourages wide-ranging debate on beneficial water uses, costs, benefits, etc.

The Province of Manitoba has been divided into nineteen watersheds (Figure 5). At present, only the Souris (7), Red (8), Grass-Burntwood (12) and part of the Saskatchewan (4) rivers watersheds have been classified.

FUTURE PLANS

The Manitoba Surface Water Quality Objectives provide the principal foundation for protection and management of water quality within Manitoba. It is essential that the program continue to evolve in order to incorporate new scientific findings related to specific chemicals, new ecological principals related to methods of offering protection to aquatic ecosystems, as well as innovative administrative procedures in order to effectively use the objectives for their intended purpose.

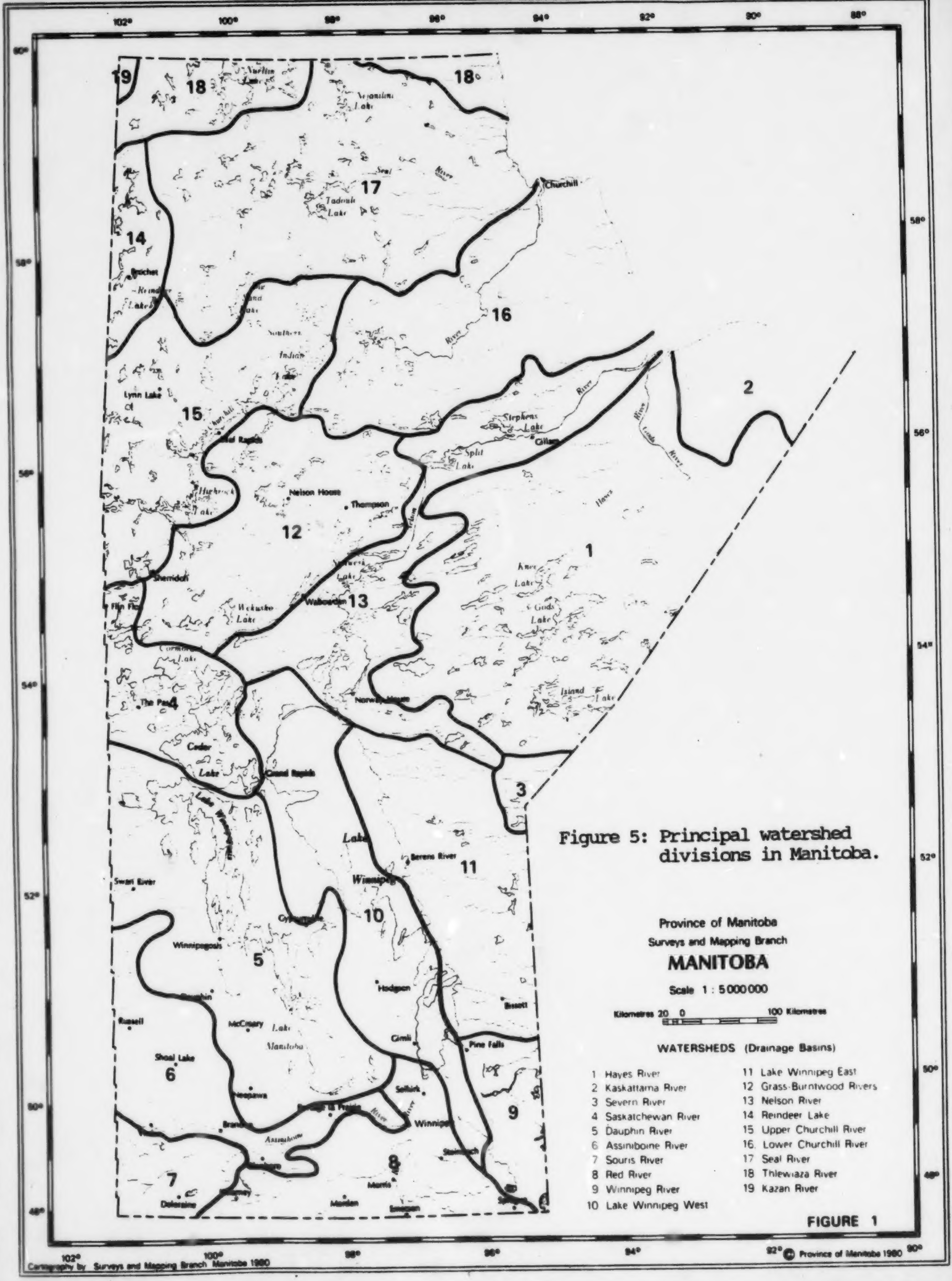

Figure 5: Principal watershed divisions in Manitoba.

A number of new and continuing initiatives are being pursued at present. These are as follows:

(1). **FURTHER REVISIONS TO THE MANITOBA SURFACE WATER QUALITY OBJECTIVES:** Some recent research and assessment by other agencies has led to the possibility of adopting a number of objectives for materials not presently covered by the Manitoba Surface Water Quality Objectives. These include aluminum, selenium, total suspended solids, glyphosate, atrazine, and carbofuran for the protection of AQUATIC LIFE AND WILDLIFE and atrazine, bromoxynil, dicamba, diclofop-methyl, glyphosate, simazine, triallate, plus others for the protection of water used for DOMESTIC CONSUMPTION. This information will be evaluated in detail in the near future and recommendations will be made concerning appropriate objectives for incorporation into the Manitoba Surface Water Quality Objectives.

(2). **MANITOBA SURFACE WATER QUALITY OBJECTIVES REGULATION:** Work is proceeding on the development of a Manitoba Surface Water Quality Objectives Regulation under the Environment Act. The intent of this regulation is to provide legislative authority to the objectives, while still maintaining the necessary flexibility. The regulatory aspect would require that developers and the licencing departments be able to demonstrate either that the Manitoba Surface Water Quality Objectives would not be exceeded by the project under consideration, or in limited cases where exceedences may be allowed, that such exceedences would not lead to the impairment of a water use. If demonstration of either of these is not possible, sufficient safeguards must be provided in the Licence to ensure that the objectives would not be exceeded as a result of that specific development. All of the narrative and numerical surface water quality objectives will be contained in schedules attached to the regulation, along with specific watershed classifications.

(3). **IMPLEMENTATION GUIDE:** The success of the Manitoba Surface Water Quality Objectives programs relies upon the ability of all planners, developers, industries, municipalities, consulting firms, plus various government agencies to completely understand the intended use of the objectives. To this end, the development of an implementation guide is being planned, that would provide explicit examples, complete with the necessary detail and complexity, for the most common point-source and diffuse-source applications.

(4). **WATERSHED CLASSIFICATION:** WATERSHED CLASSIFICATION is continuing in a number of areas of Manitoba. The regions affected by hydro-electric development in northern Manitoba are presently being classified. Water quality management plans, complete with monitoring, assessment, and classifications are being developed for individual Watershed Conservation Districts. Additionally, WATERSHED CLASSIFICATION will be used to assist in the resolution of water quality concerns on the Red and Assiniboine rivers within and downstream of the City of Winnipeg.

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Development and Use of Water Quality Objectives in British Columbia

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INTRODUCTION

The development of water quality objectives in British Columbia by the Ministry of Environment began in 1982 in response to recommendations by the Auditor General that the Ministry develop the means to measure its performance in safeguarding environmental quality. The system that has evolved features the development of province-wide water quality criteria (guidelines) which are used along with local information to develop water quality objectives for specific waterbodies. The first criteria and objectives produced by this system were approved by the Ministry in 1985 and monitoring to determine the degree of attainment of the objectives began in 1987. As of 1989, we have criteria approved for 10 contaminants or classes of contaminants* and objectives for 23 waterbodies, ranging from small lakes and streams to major lakes, segments of major rivers, and coastal marine/estuarine waters. Similar numbers of criteria** and objectives are in preparation. About $1 million/year is currently being spent by the Ministry of Environment to develop and monitor for water quality criteria and objectives, including about six person-years in headquarters for preparing criteria and objectives, overseeing monitoring, and assessing the extent to which objectives have been achieved.

DEVELOPMENT OF WATER QUALITY CRITERIA

The water quality criteria being developed in B.C. are analogous to the Canadian Water Quality Guidelines (CCREM, 1987). They are the levels of physical, chemical or biological characteristics of fresh or marine water, sediment or biota which will protect specific water uses (drinking, recreation, aquatic life, irrigation, livestock watering, wildlife, industrial water supply). The criteria are province-wide policy guidelines used for the assessment of environmental quality and are a key factor in the development of water quality objectives. We have adopted the Canadian Water Quality Guidelines (CCREM, 1987) as working criteria to be used until superseded by approved B.C. criteria for a given contaminant.

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* Particulate matter (suspended solids, turbidity, bottom sedimentation), nutrients and algae (phosphorus and chlorophyll a), cyanide, molybdenum, nitrogen (nitrate, nitrite, ammonia), copper, lead, microbiological indicators (coliforms, etc.), aluminum, and mercury.

** pH, PCB's, chlorine, fluoride, chlorophenols, dissolved oxygen, cadmium, and colour.

The contaminants chosen for criteria development in B.C. are those judged to be most urgently needed for water quality objectives development, using a ranking system that considers the number of waterbodies where objectives are needed for a contaminant, the relative toxicity of the contaminant, and the perceived adequacy of existing (working) criteria from other jurisdictions.

The criteria are prepared via an in-depth review and analysis of the toxicology literature and criteria from other jurisdictions; new research is rarely done. The result is criteria that are more thorough and comprehensive than the Canadian Water Quality Guidelines (CCREM, 1987), and more specific to B.C. conditions. The criteria reports are reviewed by provincial and federal government health and environment agencies before being finalized, and are subject to review and revision as new information becomes available.

To the extent that information is available, we try to develop criteria for fresh and marine waters, and for the water column, biological tissues, and the bottom sediments, as appropriate. Our aquatic life criteria for water generally follow a modification of the EPA model with average and maximum criteria based on long-term no-observed-effect-levels and short-term toxicity tests, respectively, for the most sensitive aquatic species in B.C. The application of average criteria requires more frequent monitoring than has been traditionally used, but yields a more detailed picture of the conditions in the environment.

DEVELOPMENT OF WATER QUALITY OBJECTIVES

Water quality objectives are criteria adapted to protect the most sensitive designated water use in a specific waterbody, taking local circumstances into account. Candidate waterbodies for water quality objectives are normally identified by the regional offices of the Ministry of Environment. These are waterbodies with existing or potential water quality problems that are deemed to be most in need of objectives. The objectives are normally prepared by headquarters staff with input from regional staff.

A water quality assessment of the waterbody is conducted which considers the present and potential water uses, waste discharges (point and non-point), water flow and movement, the quality of the water, biota and sediment, and the available water quality criteria. Water quality assessments are conducted using existing data whenever possible but, if there are significant data gaps, short-term monitoring programs are conducted. The assessment identifies the water uses that should be protected, the existing and potential quality of the water and the causes of any impairment, and the contaminants for which objectives are needed.

Objectives for water, sediment or biota are then developed for these contaminants to protect the most sensitive water use designated for protection using the available approved or working water quality criteria. The criteria may be applied as is, or modified to account for local conditions that are different than those upon which the criteria are based (e.g. water chemistry, species or life stages, water treatment used, etc.). A report is prepared documenting the water quality assessment, the designated water uses, the water quality objectives, any water or waste management actions needed to achieve the objectives, and monitoring programs needed to determine whether the objectives are being achieved or to fill data gaps for the development of future objectives.The report is reviewed by provincial and federal government health and environment agencies, and the local governments, industries, and resource development agencies involved prior to being finalized. The objectives are subject to review and revision as new information becomes available.

Water and sediment objectives apply everywhere in a waterbody except in the initial dilution zones of waste discharges, which are zones usually stretching up to 100 m from a discharge, but not exceeding 25 to 50% of the width of the waterbody, where the initial mixing of the waste and the water occurs. Objectives for biological tissues also apply within initial dilution zones to avoid harmful bioconcentration. Objectives do not apply beyond the extreme high (e.g., 1-in-10 year flood) or low flows (e.g., 1-in-2 to 10 year low flows) used for the design of waste treatment facilities.

USE OF WATER QUALITY CRITERIA AND OBJECTIVES

Water quality criteria are used by the Ministry of Environment:

(1) to provide policy guidance for the assessment of water quality and water quality impacts, and project design in areas where objectives are not yet established.

(2) as one of the two key policy inputs for the development of objectives (the other key policy being the water use designation).

Water quality objectives provide policy direction to environmental managers in the regulation of water, wastes, and other activities affecting water quality. Objectives and criteria have no legal standing in B.C., but they provide policy direction or guidance for the preparing of permits, licences, orders, and plans which are the legal instruments used to regulate activities affecting water quality.

While the development of water quality criteria and objectives and the monitoring to determine if the objectives are being achieved is the responsibility of the Water Management Branch, the use of the criteria and objectives lies mainly with the Waste Management Branch. In the regulation of wastes, they consider the water quality criteria or objectives, the provincial Pollution Control Objectives (Ministry of Environment, 1977, 1979) for effluent quality, and the available waste treatment technology to determine the appropriate quantity and quality of effluent to be specified in Waste Management permits. Generally, the most stringent of these three factors determines effluent permit limits. Back-calculation from the effluent to receiving water quality is used to ensure that receiving water quality objectives or criteria will be met under worst-case conditions. If they are not, improved waste treatment is sought, and effluent permit limits are tightened to ensure that the water quality objectives or criteria will be met. In our experience, the existence of water quality objectives and criteria have become a driving force in improving effluent treatment and quality.

MONITORING FOR OBJECTIVES

Wherever objectives are established, monitoring is conducted to determine whether the objectives are being achieved. This monitoring began in 1987, and about $0.7 million/year is currently spent to check objectives and to gather data for new objectives. Monitoring to check objectives typically features weekly monitoring for one or more 30-day periods (i.e., 5 samples in 30 days) per year during the critical periods when the objectives are most likely to be exceeded (e.g. low flows). Initially, objectives are being checked for 3 successive years before deciding whether to decrease the frequency of checking.

A report is prepared annually on the degree to which objectives were achieved in the previous year and this provides feedback to the environmental managers and the Ministry executive for taking remedial action where objectives have not been achieved. These reports are expected to be a key input to state-of-the-environment reporting. In 1987, monitoring showed a 90% success rate in achieving the objectives established to that time.

NEW DIRECTIONS

Some new directions for water quality criteria and objectives that we would like to see initiated or expanded include:

Increased cooperation between federal, provincial and territorial agencies for objectives development for waters where there is mutual jurisdiction such as transboundary waters. We have recently begun some initial projects with federal and provincial agencies and hope that mutually agreed upon objectives can be developed cooperatively.

Increased cooperation between federal, provincial and territorial agencies for sharing information and research costs for criteria development.

Increased effort on the development of criteria and objectives for sediments, biota, and the marine/estuarine environment

Increased use of site-specific toxicity testing for the development of objectives.

Field verification of objectives to determine if they are adequately protecting the aquatic environment and its uses.

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*Similar Pollution Control Objectives also govern Municipal Type Waste Discharges (1975), The Chemical and Petroleum Industries (1974), and Food-processing, Agriculturally Orientated, and Other Miscellaneous Industries (1975).

Federal Approach to Water Quality Objectives for the St. Lawrence River

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INTRODUCTION

With a drainage area of 1 180 000 km^2, the St. Lawrence is one of the most important rivers in North America. Its flow, of 6500 m^3.sec^{-1} at the exit of Lake Ontario, reaches 11 400 m^3.sec^{-1} near Quebec City, some 400 km downstream. It also drains one of the most industrialized regions of the continent, the Great Lakes, and, therefore, receives a large part of its contaminants from upstream. However, in Quebec province, the river also receives input from important urban and industrial regions, and from agricultural activities.

Some of the river's contamination problems are well known by the public: for example, the endangered species status of the beluga population, which is highly contaminated by organic chemicals. Water, sediment and fish are also contaminated in most parts of the river. Hence, water quality objectives are needed as part of the clean-up program for the river which is to restore the beneficial uses of the water. At the beginning of the 1980's, the Inland Waters Directorate decided to participate in the establishment of water quality objectives in various parts of Canada. It is only recently that the Quebec Region has had the opportunity to participate in this activity. The development of objectives has been included in the Protection component of the St. Lawrence Action Plan, which was designed to restore the river's quality and its beneficial uses.

This five-year Action Plan was implemented in June 1988, with the objective of reducing, by 1993, about 90 percent of the liquid toxic waste discharged into the St. Lawrence by the 50 industrial plants considered to be the river's major polluters. Because of international and interprovincial implications, this Action Plan is of national interest and involves several federal departments, such as Fisheries and Oceans, Industry, Science and Technology, and Environment. Environment Canada is the lead agency, with the Quebec government cooperating through the participation of the Ministère de l'Environnement and the Ministère du Loisir, de la Chasse et de la Pêche.

The St. Lawrence Action Plan has four related components. The PROTECTION part deals with the reduction of the amount of toxic substances discharged directly into the river by the 50 industries, as well as with the evaluation of the input of toxic substances from the Great Lakes, the international portion of the river and the main tributaries within Quebec province. The second part, ENVIRONMENTAL TECHNOLOGY, deals with the development of waste treatment technology to assist industry in its clean-up programs; it also has a laboratory division which develops appropriate ecotoxicological analyses for the characterization of effluents and toxic effects in the environment. The RESTORATION part deals with the clean-up of federal sites and the restoration of wetlands. The last part, CONSERVATION, includes the creation of a marine park at the mouth of the Saguenay, the protection of endangered species, the conservation of habitats and the improvement of the knowledge about the state of the St. Lawrence ecosystems.

The St. Lawrence Centre has been created to give support to the scientific, development and management activities of the St. Lawrence Action Plan. Its operational structure is composed of three divisions, ECOTOXICOLOGY AND ECOSYSTEMS, KNOWLEDGE OF THE STATE OF THE ENVIRONMENT, and TECHNOLOGICAL DEVELOPMENT. The Toxic Inputs Section of the Ecotoxicology and Ecosystems Division is responsible for the development of water quality objectives for the St. Lawrence River as well as for the evaluation of input, sources and mass balances of priority pollutants, and the evaluation of the fate of toxic substances in water, sediments and organisms. These activities were initiated in 1989 by a pilot study in Lake St. Francis, and included water sampling, sediment sampling (bottom and suspended), sediment profiling, and biological studies using phytoplankton and zooplankton bioassays, bioaccumulation in young-of-the-year fish, in mussels and in macrophytes, and enzyme analyses (MFO). These studies will help to establish long-term programs regarding sampling, analytical and statistical methodology.

DEVELOPMENT OF OBJECTIVES

This activity is to produce objectives for the quality of the river as an ecosystem, with respect to the presence of toxic substances and to quantify the improvements of the river quality following the St. Lawrence Action Plan interventions.

In this process, the first steps are to collect and summarize the information on the river's uses, quality and pollution, and to elaborate a process to select criteria for the development of the water quality objectives. In more details, for the current year (1989-90), we plan to:

- collect and summarize information on the river's uses, quality and pollution;
- elaborate a list of priority substances (causing pollution);
- elaborate a list of priority sources of pollution;
- select hydrozones and priority interest zones;
- form a committee or working group for the development of water quality objectives;
- select criteria and elaborate a process for the development of water quality objectives;
- develop water quality objectives for Lake St. Francis, and
- develop preliminary water quality objectives for the St. Lawrence River.

A preliminary list of priority substances has been established, based on the criteria of presence and toxicity, as well as their inclusion on the CEPA list of priority substances (Table 1). This preliminary list will be the object of a more detailed evaluation to select the final priority substances list. The list of priority sources will be based on a contract report that identified the most important tributaries in terms of toxic input. The hydrozones concept is based on homogeneous zones of the river from a hydrodynamic point of view. As the St. Lawrence River is made up of a succession of lacustrine and fluvial zones, these hydrozones will serve as a basis for our sampling program. The "Priority Interest Zones" is another concept developed by the St. Lawrence Action Plan that will allow the choice of a few zones in the river as regions with specific and acute pollution problems, impaired uses or conservation objectives; St. Lawrence Action Plan activities will be concentrated in these zones, and after intervention, monitoring will hopefully show significant improvements to the public.

A working document will be produced, including information on the various subjects mentioned above, and will be sent to the members of the working group. They would then participate in a workshop to discuss the document and develop preliminary water quality objectives. These would be developed first for Lake St. Francis, since it is the first section or hydrozone of the river in the Quebec region and there are no major Quebec sources of contaminants after the river leaves the Cornwall-Massena area. Since a few water quality objectives have already been developed for the Great Lakes, this will aid the process of developing local ones.

Afterwards, water quality objectives will be developed for the remainder of the river, using the hydrozones approach, which takes into account the changing features of the river. A monitoring program will be set up and an implementation plan elaborated for compliance monitoring. Complementary studies will be made where necessary and objectives will be reevaluated in the view of new data and updated knowledge. The objectives will be proposed to concerned governments for adoption and will serve in monitoring to detect river quality improvement. Development of sediment and ecosystem objectives is a more long-term goal, since guidelines have not yet been developed.

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Table 1. Priority substances list for toxic input studies

SUBSTANCES	IN ST. LAWRENCE RIVER			TOXICITY	PRESENCE ON CEPA LIST
	W	S	O	(2)	
As	+	+	+	***	yes
Cd (1)	+	+	+	***	yes
Cr	+	+	+	**	yes
Hg (1)	+	+	+	***	yes (ann.1)
Ni	+	+	+	***	yes
Pb	+	+	+	***	yes (ann.1)
Se	+	+	+	**	no
V	+	+	+	***	no
Aldrin-dieldrin (1)	+	+	--	****	no
Chlordane	+	+	+	***	yes (pp)
DDT (1)	+	+	+	***	no
Endosulfan	+	+	--	****	yes (pp)
Endrin	+	+	--	****	no
HCB	+	+	+	***	yes
HCH	+	+	+	****	no
Lindane	+	+	--	****	no
Methoxychlor	+	+	--	**	yes (pp)
Mirex	+	+	+	***	yes (ann.1)
PCB (1)	+	+	+	**	yes (ann.1)
PAH	+	+	+	**	yes
Benzo(a)pyrene	+	+	+	*	yes
12-dichlorobenzene	+	--	--	***	yes
14-dichlorobenzene	+	+	--	**	yes
124-dichlorobenzene	+	+	+	***	yes
2,3,7,8-TCDD	--	--	+	****	yes
Di(ethyl-hexyl)phthal.	+	--	--	***	yes
Dioctyl phthalate	+	--	--	***	yes
Trichloroethane	+	--	--	*	yes
Trichloroethylene	+	--	--	*	yes
Tetrachloroethylene	+	--	--	**	yes

(1) concentrations in St. Lawrence River water are greater than the CCREM (1987) guidelines.

(2) * low toxicity;
** LC50 < 10 mg/L;
*** LC50 < 1 mg/L or LD50 < 500 mg/kg;
**** LD50 < 50 mg/kg.

from GLWQB 1982 and Laliberté et al. 1985.
+ = presence in W (water) S (sediment) and O (organism).
-- = not present or no data found.
CEPA Canadian Environmental Protection Act.
(ann.1) substance is listed on appendix 1 of CEPA.
(pp) Priority pesticide (Cossette et al. 1988).

Approaches to Estuarine and Coastal Sediment Quality Objectives

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INTRODUCTION

In order to discuss a wide variety of environmental contaminants for which sediment quality objectives are being considered, we often categorize chemicals with similar characteristics; hence: organic vs. inorganic, conventional versus"toxic", persistent versus biodegradable, and so on. Mackay (1988) defined three categories of environmental contaminants with a different perspective:

- Disruptives: nonselective or largely narcotic effects because of their sheer quantity, and not because of any subtle biochemistry. Examples are salt, suspended solids and some trace metals. For liquids, the LC50 is typically 5 to 20% of the solubility in water. For these, toxic thresholds are easily measureable and unequivocal.

- Distributives: chemicals with unusual biochemical activity or partitioning characteristics. Examples are mercury, low-molecular weight polycyclic aromatic hydrocarbons (PAHs) PCBs and DDT. For these, the definition of effects thresholds begins to break down. For example, effects thresholds for DDT were well known and farmers calculated application rates carefully to kill insects and avoid spending too much on the expensive insecticide. At levels far below the acute effects thresholds, however, DDT residues continue to kill predacious birds by eggshell thinning many years after banning. For these chemicals, we need to think about "ecosystem effects" that go beyond toxic thresholds, and recognize that persistent and strongly bioaccumulative contaminants can be a hazard at any level.

- Directives: chemicals whose molecules have the capacity to direct the future well-being of an organism. Mackay (1988) gave the example of a virus that exerts control out of all proportion to its mass. Examples are high molecular weight PAHs and Ra226, which are known carcinogens. Current theory is that effects are possible at any level, and risk increases with exposure. For these, it may not be appropriate to set sediment quality objectives other than zero. This paper examines equilibrium partitioning and effects-based approaches to setting sediment quality objectives to assess their applicability to these classes of pollutants.

EQUILIBRIUM PARTITIONING

Any serious discussion of sediment quality guidelines in Canada has to start with the fugacity models of Don Mackay and his colleagues at the University of Toronto (Mackay and Paterson 1981, 1982). These models calculate the partitioning of chemicals into air, water, sediment and biota, based on their measureable physical properties such as octanol-water partition coefficients (Kow), vapour pressures, Henry's rate constants, etc. The idea is that the partitioning model is used to predict what the water concentrations would be in interstitial water, given the concentrations in sediment. It is also used to establish site specific objectives based on water quality guidelines developed from bioassays exposing animals to contaminants in an aqueous medium. As such, they really only address contaminants in Mackay's (1988) "distributive" category.

Equilibrium partitioning is an attractive approach because the input data are simple measurements of physical properties of the chemical, These are readily reproducible in different laboratories and do not have the large animal species variability with which toxicologists have to deal. However, even equilibrium partitioning proponants note problems with the procedure (Shea 1988):

- Both synergistic and antagonistic interactions occur among contaminants, but given our lack of knowledge in this area, there is no way to presently incorporate these effects into the equilibrium partitioning model.

- Contaminant accumulation by direct ingestion can be a significant source, highly dependant on the organism and its local environment. This is particularly true of marine environments, where many detritivores are important both ecologically and commercially as harvested for human consumption.

- For many contaminants, water quality guidelines are based on a limited toxicological data base. Therefore, any sediment quality guidelines using these water quality guideline values may be inadequate to ensure protection of those life forms not included in the original toxicity studies, and this is particularly true for the marine environment.

- There is a lack of chronic data for many organic contaminants, and existing data, for both metals and organic compounds, are based on total aqueous concentrations, not on a single contaminant species.

- For contaminants with very high Kow's, partitioning is by hydrophobic exclusion from bulk water, rather than association reactions at the sediment-water interface. Therefore water quality guidelines are not appropriate and "effects thresholds" need to consider bioaccumulation in individuals and effects on ecosystems.

- A decrease in partitioning efficiency has been observed for sandy or coarse grained sediments.

- Polar organic compounds may adsorb on sediments by a variety of mechanisms; unfortunately, there is currently no means of modeling the adsorption of polar organic contaminants as a class. Instead, partitioning models have been compound-specific and have not had much success. In addition, the complicated surface chemistry of polar organic contaminants precludes the possibility of developing sediment quality guidelines for these compounds in the near future.

- In the marine environment, precipitation of metal sulfides can be an additional complexity because many coastal and estuarine regions have only a thin oxidized layer above anoxic sediments.

To the above drawbacks reviewed by Shea (1988), I would add the comment that equilibrium partitioning is not appropriate to Mackay's "directive" contaminants, which exert control over organisms out of all proportion to their mass.

These difficulties have lead many investigators to examine biological effects-based procedures as an alternative to establishing sediment quality guidelines (cf. Tetra Tech Inc. 1986; Chapman 1989).

APPARENT EFFECTS THRESHHOLDS

Several methods are available for deriving biological effects thresholds from toxicological data or from synoptic measurements of physical, toxicological data, as reviewed by Chapman (1989). Chapman (1986) and Chapman et al. (1987) proposed a sediment quality "triad", which used bulk sediment chemistry, sediment bioassay and bottom fish tissue abnormalities. Neff et al. (1987) gave a "screening level" approach that used bulk sediment chemistry and infaunal community data. Along the same lines, Schwartz et al. (1985) derived a "no-effect" concentration for acute toxicity of total PAH using amphipod (Rhepoxynius abronius) toxicity data from Eagle Harbor, Washington.

Tetra Tech Inc. (1986) examined several approaches, including equilibrium partitioning and "Apparent Effects Thresholds" (AET), which are the levels of contaminants above which statistically significant, adverse biological effects were always observed. They used synoptic (i.e. samples were collected at the same time and place) chemical, sediment bioassay and benthic community data from 104 stations in Puget Sound to derive AETs for oyster (Crassostrea gigas) larva bioassays, amphipod (Rhepoxynius abronius) bioassays, benthic community analyses and luminescent bacteria (Photobacterium phosporeum) bioassays. The AET values have since been updated with new bioassay and community data from 338 stations (Barrick et al. 1988). AETs result in higher values than guidelines based on the lowest observed effect level. The method eliminates the possible confusion over which contaminant(s) is (are) causing the observed effects at a particular station by only using stations at which no effects were observed to establish the guidelines. Conversely, at stations with levels of a contaminant above the no-effect level, effects were always unequivocally demonstrated. If effects were observed at lower levels elsewhere, they were assumed to have been caused by other contaminants or natural conditions.

The AET approach is being considered as the basis for legal marine sediment quality standards in the States of California and Washington (WSDE 1988) and is being evaluated by the U.S. EPA for national application.

Of the various approaches to derive effects-based sediment quality guidelines, the AET method is clearly the most mature scientifically. It uses unequivocal effects criteria (LC_{50} for bioassays and a 50% reduction in species abundance of major taxa for benthic community analysis with 95% confidence intervals). It has been applied to many chemicals, including all 129 EPA Priority Pollutants, with numeric guidelines given for each of the above-listed biotests. Originally developed for Puget Sound Dredge Spoil Analysis, the approach and resultant values have been adopted by the Washington Department of Ecology, Region 10 EPA, and the Puget Sound Water Quality Authority.

Yet the AET approach is not without problems:

- First, the three bioassays are all short-term, acute tests. They do not account for more subtle or longer acting effects, such as the physiological and metabolic effects on benthic fish and invertebrates caused by sediment contamination with organochlorines from bleached pulpmill effluent (see review by Colodey 1989). They also do not specifically address "directive" chemicals as defined by Mackay.

- The second problem is that the AET approach, based solely on acute bioassays and benthic community impacts, does not account for effects of highly bioaccumulative contaminants, such as DDT and PBCs, for which ecosystem effects may be more serious than individual toxic effects. (The benthic community test only passes if it contains an abundance of major taxa rigorously analysed, i.e. a reasonably mature community; hence, it may be thought to account for ecosystem effects. The taxa involved, however, are not necessarily at top trophic levels, and could at any rate be replenished by recolonization and immigration in the required time scales to pass the tests).

- The third difficulty I have with the AET approach is that it apparently assumes similar bioavailability of trace metals throughout the region (apparently, because this assumption is not explicit in the description of the procedure). Toxicity of trace metals varies with bioavailability, rather than amount, of the metal. Trace metal bioavailability in marine systems varies with a myriad of factors, including geochemistry, the chemical species of the source contaminants, redox potential, grain size and organic carbon content of sediments. In general, however, trace metals are rarely very bioavailable in marine systems. A corollary is that the chance of sampling a station with high metal bioavailability increases with the number of sampling stations.

To illustrate this, Table 1 shows the Lowest (among the four biotests) Apparent Effects Threshold (LAET) values for a number of trace metals based on 104 stations in Puget Sound (Tetra Tech. 1986) versus the new LAETs based on an expanded data set of 338 stations and a proportional number of new bioassays (WSDE 1988). The new LAET values are nearly all different from, and mostly quite higher than, the old LAET values. The possibility exists - indeed, I feel is quite strong - that the earlier values were more representative of the "usual" degree of bioavailability, and hence toxicity, and that the increase in sample size increased the probability of sampling stations with anomalously high bioavailability at higher levels, which resulted in dangerously higher LAETs.

Table 1. AET: 1986 vs 1988 Data Sets

Trace Metals

Chemical	Old LAET	New LAET
Antimony	3.2	150
Arsenic	85	57
Cadmium	5.8	5.1
Chromium	27	260
Copper	310	390
Lead	300	450
Mercury	0.41	0.41
Zinc	260	410

This same problem does not exist with organics, as bioaccumulation rates do not change as radically with local conditions, as illustrated in Table 2: New LAETs were generally the same as the old LAETs. An exception is PCP, for which an AET value had been impossible to establish in the earlier data set because no effects were demonstrated unequivocally at any station.

Table 2. AET: 1986 vs 1988 Data Sets

Organics

Chemical	Old LAET	New LAET
Total PCB	130	130
LPAH	5200	5200
HPAH	12000	12000
PCP	>140	360
Phenol	420	420
Dibenzofuran	540	540
Dimethylphthalate	71	71

- Finally, the fourth major reservation I have about the AET approach is that it contains no inherent safety factor. Since the criteria are set at the levels above which effects were always observed, the approach virtually guarantees damage to marine life if pollution reaches as high as the given guidelines. Tetra Tech. Inc. (1986) recognized this, and gave a more conservative list of guidelines based on a statistical procedure that assumed an error in the designation of "no effects", similar to selecting the lower confidence limit, rather than the central value.

Similarly, the State authorities applied a safety factor of 0.1, based on the "usual" acute:chronic ratio of 10:1 for bioassay test results for a variety of contaminants (WSDE 1988).

The problem I see with this safety factor is that it does not seem appropriate for Mackay's "directive" category of contaminants, for which acute:chronic ratios may be very much higher; nor for the more bioaccumulative of his "distributive" chemicals that may have ecosystem effects far more serious than individual effects based on acute bioassays and simple community analyses.

SOLUTIONS

Equilibrium partitioning is appropriate for some categories of contaminants. These include most of the "distributives", although caution is indicated for those with high Kows (e.g. PCBs). Equilibrium partitioning is not indicated for metals (or metalloids) and ionic or polar organic compounds; or for compounds without a solid toxicity database.

Good candidates for sediment quality objectives using the equilibrium partitioning approach are chlorophenols and some individual constituents of pulpmill effluents, including dioxins, as "tracers" for whole effluent. Indeed, Suntio et al. (1988) have listed the available data on physical properties of some 250 constituents of pulpmill effluent needed as input to Mackay's fugacity models, and have added biological data such as bioconcentration potential and toxicity. They do make the point, however, that given the absence of input data for some of the constituents, and the possible cumulative rather than additive toxicities, methods are needed to treat these chemicals as a class, rather than on an individual basis. It may be a useful strategy to use the equilibrium partitioning approach for these until an acceptable database of toxicity is developed, and then use an effects based procedure.

The equilibrium partitioning approach cannot, of course, model all the toxic constituents of pulpmill effluents, many of which have not even been identified, let alone described in terms of physical properties (Colodey 1989). Cumulative effects of bulk sediments contaminated with pulpmill effluent could, however, be assessed, and quality guidelines derived, using the AET approach if it were modified to include longer term bioassays and to assess physiological effects in benthic species (cf. Colodey 1989) in addition to benthic community analysis. This would require a considerable resource commitment.

Physical data are also available for 2,3,7,8 tetrachlorodibenzodioxin (Podoll et al. 1986), as are toxicity data for some species (cf. Colodey 1989) as well as limits in fish tissue for human consumption set by Health and Welfare Canada. An example of this process would be to use a partitioning model such as Mackay's fugacity models to calculate the levels in sediment needed to meet levels in edible fish and invertebrate tissues of 20 ppt set by HWC, and the levels in water (the target is interstitial water) of the lowest known toxicity threshold of 38 pg/L (parts per quadrillion) to rainbow trout exposed for 28 days (Mehrle et al. 1988), and divide by an appropriate safety factor.

Trace metals have rarely accumulated to harmful levels in marine food webs in Canada, although exceptions that resulted in temporary closure of important fisheries on both the east and west coasts are notable. We already have sediment standards for trace metal content in materials proposed for Ocean Dumping, and good controls for trace metals in land-based effluent. Consequently, I would recommend a low priority for development of uniquely Canadian sediment quality guidelines for trace metals. Instead, I would recommend using existing guidelines developed by other jurisdictions for the time being, such as the AET levels for trace metals developed for Puget Sound. We have, in fact, been using AET levels to assess in situ sediment contamination in the Pacific & Yukon region for two years.

For some high Kow "distributives" such as PCBs, I believe the AET method needs to be refined to incorporate into the model the high bioaccumulation potential that reflects a broader ecosystem concern than merely toxicity to some organism. A colleague and I have proposed a method to do this elsewhere (Harding and Waters 1989). This method, as applied to PCBs, involves simply dividing the LAET of 130 ppb by the log of the bioconcentration factor (BCF), or by its correlate, log Kow-1.32, given by Mackay (1982). This method produces a PCB sediment quality guideline of 26 ppb. Interestingly, this is very close to the British Columbia provincial sediment quality objective for the Fraser River Estuary of 30 ppb, and the proposed CCME (CCREM) value of 50 ppb.

For "directives" such as HPAHs, I believe the AET method as currently developed is fundamentally flawed. This is because none of the bioassays used in the procedure test for carcinogenicity, a major concern of HPAHs. These compounds are not very acutely toxic, and the AET using only acute bioassays gives a dangerously high result. As noted above, the nominal acute:chronic ratio of 10:1 is also not validly applied to HPAHs, whose acute:chronic ratios are much higher. For these compounds, a more realistic (as well as practical) approach would be to simply note the sediment HPAH levels associated with abnormal rates of liver lesions, as provided by Goyette and Boyd (1989) for Vancouver Harbour, and set the sediment quality guideline lower by an appropriate safety factor (again, the BCF or log Kow-1.32 is suggested). This would result in sediment quality objectives approximately 10% of those that would be established using the LAET modified by the acute:chronic ratio.

Another alternative would be to re-do the AET procedure, adding a genetic damage test such as chromosome aberration, sister chromatid exchange or the Ames test for mutagenicity. A currently planned study of synoptic chemical analysis, bioassays and infaunal community analysis in Vancouver Harbour would enable such calculations.

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Canadian Council of Ministers of the Environment

Workshop on the development and use of water quality objectives
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